THE IMPLICATIONS OF FYNBOS PHYTOGEOGRAPHY FOR WETLAND PHYTOASSESSMENT.

By

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Declaration:

I know the meaning of plagiarism and declare that all the work in the document, save for that which is properly acknowledged is my own.

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Abstract

Wetlands are recognized as being important ecosystems supporting considerable biodiversity and providing essential services that maintain the condition of surrounding environments as well as providing resources directly utilized by humans. The plants that grow within wetlands exist as a result of the environmental stresses and opportunities within the habitat in which they occur and hence aspects of the vegetation have proven useful as indicators of these environmental constraints. Many human landuses and activities impact negatively on the environmental conditions within wetlands, altering the present environmental state and impairing the potential to support a floristic composition equivalent to unimpaired wetlands. An indication of the environmental condition of a given wetland can therefore be provided by a comparison of the species composition, diversity and functional organisation of plants within that wetland relative to those of minimally impaired wetlands within the same region.

The initial focus of the present study was intended to develop phytoassessment tools for the evaluation of the environmental condition of lowland wetlands in the Fynbos biome of the South Western Cape of South Africa. Recent meta-analysis of the limited baseline phytosociological data for wetlands in the South African context suggested broad azonal distribution of wetland plants, with hydrodynamics and/or salt content as azonality-driving macro-ecological factors. In contrast climatic and edaphic factors are considered to drive the distribution of terrestrial (or dryland) vegetation into zones or zonobiomes. For the purposes of wetland phytoassessment, therefore, the south-western Cape coastal lowlands was considered as a single and homogenous wetland phytogeographical region within the Fynbos biome of the Cape Floristic Kingdom. A field survey and analysis of freshwater wetland vegetation across the Cape coastal lowlands (below 200m a.s.l.) was undertaken in the present study to explore the potential to identify aspects of the vegetation that could serve as indicators of the present environmental state within wetlands. The Braun Blanquet approach was employed for the vegetation sampling of 59 wetlands in three subregions of the coastal lowlands of the Fynbos biome. The consideration that the south-western coastal lowlands of the Fynbos biome represent a single region for phytoassessment was assessed. The distribution of the apparently azonal wetland vegetation was compared to that of the zonal units of vegetation within

the Fynbos biome using non-metric multi-dimensional scaling ordinations and also with constrained ordinations based on discriminant analysis. Environmental data were assessed to examine their correlation with the distribution of wetland vegetation using canonical analysis of principle components. Rainfall and temperature data were based on interpolated values at a resolution of 1' by 1' of a degree. Soil and water samples were analyzed for a wide range of physico-chemical variables. The extent and intensity of anthropogenic landuses and disturbance influences were qualitatively assessed in each wetland, and within 500m around each wetland, in order to determine the amount of anthropogenic impairment and/or relative lack of impairment.

This study revealed phytogeographical patterns that suggest that floristic composition and distribution of wetland plants is strongly influenced by the same drivers (e.g. rainfall, temperature and geology) that influence terrestrial vegetation. This contradicts earlier descriptions of wetlands as containing relatively homogenous floristic units with azonal distribution determined by hydrological regime and salinity. The climatic variation and diversity of edaphic types within the Cape coastal lowlands hinders the ability to determine phytoassessment tools with broad geographical applicability. Considerable beta diversity variation was found to exist in the 373 wetland plant species across the wetlands of the Cape coastal lowlands with a large number of species unique to each sub-region and very few (<30 species) common to all. A large contingent of graminoid and of annual taxa existed in these wetlands. A greater percentage of annual taxa were evident in the wetlands from the north-west than the central and south-eastern sub-regions of the Cape coastal lowlands. Distinctly different phytosociological units of wetland vegetation were found to exist in each of the three sub-regions of the south-western coastal lowlands. Determination of different vegetation types and indicator species was considered beyond the scope of the present study. Within each of these sub-regions the distribution of wetland vegetation was found to mirror the distribution of different zonal (or terrestrial) units of vegetation. Rainfall and edaphic parameters (especially soil pH) correlated with (1) the phytogeographic disjunctions observed to exist between the wetland vegetation contained within each of the three sub-regions (e.g. inter-sub-regional differences); and (2) within sub-regions (e.g. intra-sub-region) between wetlands associated with each of the Strandveld, Fynbos and Renosterveld terrestrial units of vegetation. As distinct units of wetland vegetation correlated with the distribution of the zonal (or terrestrial) vegetation units this suggests that the distribution of coastal lowland freshwater wetland vegetation is therefore not azonal. Consequently the south-western coastal lowlands of the Fynbos biome cannot be considered a single region for

wetland phytoassessment purposes. The identification of aspects of vegetation that discriminate between impaired and unimpaired wetlands was found to be possible only within units of vegetation classified into naturally distinct units as based on climatic and edaphic parameters.

Across the whole data set of wetlands from the Cape coastal lowlands, and/or within the three sub-regions, insignificant difference in species complement existed between impaired and minimally impaired wetlands. Within separate units of wetland vegetation, as separated by association with distinct units of terrestrial vegetation distinctly different species and community structure existed within different zones of wetness. Within these units of vegetation separated by zones of wetness and by association with different terrestrial units of vegetation, where sufficient replicates were sampled, significantly different species were found to exist in impaired relative to minimally impaired wetlands. A number of species and other vegetation attributes were identified that occurred with discriminatory difference in impaired and minimally impaired wetlands. Greater percentage ground cover of annuals, aliens and alien graminoids occurred in impaired than minimally impaired wetlands. Greater cover of indigenous woody taxa, sclerophyllous woody taxa and indigenous leafless graminoids occurred in the minimally impaired than the impaired wetlands. The discriminatory species and vegetation attributes present the possibility of being able to perform phytoassessment of wetland environmental condition within wetlands with the same habitat from which these attributes were identified. These results suggest that vegetation attributes for phytoassessment within the Fynbos biome will need to be developed independently for different units of wetland vegetation, which at the present moment are most easily discriminated by using the terrestrial or zonal vegetation units of Rebelo *et al*. (2006).

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What would the world be, once bereft of wet and wildness? Let them be left, o let them be left, wildness and wet, long live the weeds and the wilderness yet.

from INVERSNAID *by*

Gerald Manley Hopkins

CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

1.1 Background

This thesis is based upon the work completed for a Water Research Commission project (Corry in press) researching the means of assessing environmental condition of wetlands using macrophytes, namely phytoassessment. The intent of that project was to identify and adapt established international methods of phytoassessment for use in the context of South African wetlands. The work culminated in a report describing the international history and development of bioassessment of wetlands using plants, a synthesis of the methods most applicable for South Africa and a protocol for developing phytoassessment tools. Development of non-riverine wetland phytoassessment tools have been based on the principle of comparing like habitats, differing only in degree of human disturbance, in order to be able to identify species with characteristic association with impaired relative to minimally impaired ecosystems (US EPA 2002a, Dahl 2004). The delimitation of comparable habitats to regions with similar vegetation has meant that development of local or regional wide rather than nation wide phytoassessment tools are more achievable (US EPA 2002b). Wetland vegetation of the south-western coastal lowland of the Fynbos Biome within the Cape Floristic Kingdom was used as a test case in an attempt to develop phytoassessment tools using the methods outlined internationally as best practice. This process highlighted the considerable lack of baseline knowledge about wetland plant ecology in the context of the Cape coastal lowlands and the difficulty of classification of separate units of vegetation habitat. The objective of this Masters thesis is to redress some of that lack by examining the wetland phytogeography of the Cape coastal lowlands and exploring the potential for identifying species that indicate impaired *vs* minimally impaired environmental conditions.

1.1.1 The Fynbos Biome

The south-western Cape is a unique region of southern Africa with regard to climate, geomorphology, and vegetation (Cowling *et al.* 1997). It was initially categorized as a sub-unit of the global Mediterranean biome of similar vegetation, as defined by a climatic zone of winter rainfall and hot dry summers (Walter & Box 1976). In the south-west of the Cape a significant amount of rain falls during cool winter months and is juxtaposed with summers that are hot and predominantly dry, whilst further east, but still within the biome, there is a gradient from all-year to predominantly summer rainfall. Beyond this climatic zone (outside the biome) to the north and east a summer rainfall regime exists (Cowling *et al.* 1997; Rebelo *et al.* 2006; Chase & Meadows 2007). A mosaic of geological substrates gave rise to a multitude of predominantly nutrient-poor soils in the south-western Cape (Specht & Moll 1983; Deacon *et al.* 1992; Cowling *et al.* 1997; Rebelo *et al.* 2006). The terrestrial vegetation is characterised by shrublands dominated by small-leaved (*fyn*), evergreen shrubs (*bos*) with limited lifespan (Walter & Box 1976) whose regeneration is intimately linked to fire driven nutrient cycling, reseeding and resprouting (Cowling *et al.* 1996; Rebelo *et al.* 2006). Due to the predominance of the *fyn bos* vegetation, within this mediterranean climatic zone, the south-western Cape has been named the Fynbos Biome (Rutherford & Westfall 1986, Cowling & Holmes 1992, Rebelo *et al.* 2006). The Fynbos Biome supports exceptionally high plant diversity, species richness and endemism and hence is recognized as one of the six phytogeographical Kingdoms of the world (Takhatajan 1986), named *Capensis* (after Taylor 1978 and Werger 1978) or the Cape Floristic Kingdom (CFK). Most of this endemism and biodiversity is attributable to the Fynbos vegetation unit which, together with Strandveld and Renosterveld, comprise the predominant broad terrestrial vegetation units of *Capensis* (see Rebelo *et al.* 2006 for a description of these different units). As a matter of interest, the CFK should not be confused with the Cape Floristic Region which incorporates the Fynbos Biome along with the Albany Thicket (summer to perennial rainfall) and Succulent Karroo (winter to all-year rainfall) Biomes (Goldblatt 1978, Goldblatt & Manning 2000, Rebelo *et al.* 2006).

The freshwater wetland vegetation of the Fynbos Biome is recognised as having some endemics particularly associated with Vernal Pools (described below), with shrub-dominated ericaceous wetlands, and with other Fynbos-associated wetlands, particularly those, in the mountains (Mucina *et al.* 2006a). The freshwater wetlands of the Cape coastal lowlands of the Fynbos Biome also have some genera and species with widespread and sub-cosmopolitan distribution (Keeley & Zedler 1996, Mucina *et al.*

2006a). What constitutes different units of freshwater wetland vegetation in the Fynbos Biome is, at best, broadly defined at present due to limited baseline data (see Section 1.4.4.2) (Mucina *et al.* 2006a).

1.1.2 What is a wetland?

A wetland is defined in the National Water Act of the Republic of South Africa (1998) as: "*land which is transitional between terrestrial and aquatic ecosystems, where the water table is usually at or near the surface, or the land is periodically covered with shallow water, and which land in normal circumstances supports or would support vegetation typically adapted to life in saturated soil*".

As a contracting party to the Convention on Wetlands of International Importance – the "Ramsar Convention", South Africa also adopts the Ramsar definition of wetlands: "*areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres*" (Cowan 1995).

According to the National Water Act (RSA 1998) definition all that is required for land to be termed wetland is for the substrate to be wet long enough to support plants adapted to saturated conditions. The broader context incorporated by the Ramsar Convention definition of wetlands includes many more habitat types including the water column of rivers, lakes and other freshwater basins, estuaries and shallow marine systems. The classification of different wetland habitats is defined by the National Wetland Classification System (NWCS) for the South African National Wetland Inventory (Ewart-Smith *et al.* 2006, SANBI 2009). The structure of the NWCS is initially hierarchical, with primary discriminators distinguishing between wetlands of significantly different type in regard to hydrogeomorphic and ecological character and the functions that wetlands perform (Ewart-Smith *et al.* 2006). The NWCS thus establishes a framework for research and conservation of different types of wetland and wetland habitats for South Africa. The wetlands that were the primary focus of this present thesis, and that therefore guided the literature reviewed, were freshwater inland systems within a planar landscape setting, from multiple hydrogeomorphic types and landforms, with endorheic or exhoreic drainage but a lentic hydroregime, and characterized by a range of ephemeral to permanent saturation and, if any, predominantly seasonal inundation. Rainfall and related groundwater flow were the predominant water source of these habitats but a number of

wetlands sampled were also in the floodplain of rivers and received a percentage of their water from seasonal flood events.

1.1.3 The importance of wetlands in the environment

Wetlands perform various ecosystem services that are considered to have economic and environmental value. Purification of catchment surface water, nutrient and pollutant removal, floodwater attenuation and associated sediment retention and erosion control, and groundwater recharge, are recognized environmental services that wetlands perform (Faulkner & Richardson, 1989, Johnston 1991, Reddy & Gale 1994, Richardson 1994, Costanza *et al.* 1998, Mitsch & Gosselink 2000, Zedler & Kercher 2005, Brauman *et al.* 2007). All of these wetland functions are considerably enhanced by the presence of vegetation both around and within wetlands, which tends to slow the flow of water and thus influence water quality by moderating the amount of nutrients, pollutants and sediment in aquatic ecosystems (Mitsch & Gosselink 2007). The greater the frictional resistance (roughness coefficient) offered by the vegetation the greater the ability to slow down and trap sediment and associated pollutants. Reduction of flow tends to lead to sediments being deposited on the soil surface. At the same time that sediments are trapped, other chemical constituents (*e.g.* nutrients and toxins) may also be trapped. Due to the action of anaerobic bacteria, fungi and protozoa that are present amongst plant roots and in the sediments, these chemical constituents are degraded to simpler molecules (Reddy & Gale 1994). Plants also take up nutrients and other chemical constituents, thus removing them from the substrate or the water column. Plants can thus improve water and soil quality, hence the use of vegetated man-made wetlands for amelioration of water-borne waste (Rogers *et al.* 1985).

Wetlands, particularly in arid countries such as South Africa, represent a limited habitat resource within predominantly terrestrial landscapes and are therefore a critical store of biological diversity (Ramsar COP7 1999, Williams *et al.* 2004, Dudgeon *et al.* 2006, Verhoeven *et al.* 2006, Mucina *et al.* 2006a). Furthermore, many wetlands are highly productive systems, even rivalling rainforest in biomass production and as a result have considerable economic and social value (Woodward & Wui 2001, Schuyt 2005, Brander *et al.* 2006). Conversely it is recognised that productivity is low in some wetlands because of a scarce supply of nutrients (Cronk & Fennessy 2001). In keeping with the diversity of terrestrial vegetation exhibited by nutrient-limited ecosystems such as in the South African and West Australian mediterranean floral regions (*e.g.* Kruger *et al.* 1983, Cowling *et al.* 1996), mediterranean wetlands may equally be expected to exhibit high

levels of diversity with a resultant complex mosaic of different stands of plants (community heterogeneity) in the vegetation that they support. The diversity of vegetation within wetlands of the nutrient-limited mediterranean Cape Floristic Kingdom of South Africa has not explicitly been examined (Mucina *et al*. 2006a).

The perceived value of wetlands has led to an obligation by all countries that are signatories to the Ramsar Convention to assess and monitor the ecological integrity and environmental condition of designated wetland ecosystems (IUCN 1980). In South Africa the environmental condition of wetlands needs to be ascertainable for effective implementation of the National Water Act (1998), and for a wider range of activities such as conservation planning and wetland management. The Department of Water Affairs (the national water regulation authority) identified the importance of preserving remnant wetland habitat (DWAF 2004). Wetland assessment and monitoring facilitates the sustainable utilization and conservation of these important ecosystems (Finlayson *et al.* 2002, DWAF 2004, and Malan & Day 2005a). The findings of a strategic overview of the research needs for wetland health and integrity (Malan & Day 2005a) indicated that a method for assessing and monitoring wetland biological integrity, or wetland environmental condition, was required in order to meet both national and international legislative requirements.

1.2 Human-induced impairment of wetlands

A large percentage of the world's wetland habitat, and more than half in some areas of South Africa, has been lost or severely degraded due to unsustainable agricultural, social and development practices (IUCN 1980, Breen & Begg 1989; Kotze *et al.* 1995, Shearer 1997, Dini 2004). Those wetlands that remain are highly threatened due to human population growth and development pressure (*e.g.* Kotze *et al.* 1995, Adamus *et al.* 2001). In the naturally arid conditions of much of South Africa water-stress caused by human land-use practices poses a considerable threat to much of the remaining wetland habitat. The perceived provision of ecosystem services by wetlands, and their economic value, suggests the need to sustainably manage human utilization of wetlands and their water supply and to conserve what remnant wetland habitat there is (DWAF 2004).

1.2.1 Disturbance can be natural or anthropogenic

It is important to note that disturbance to ecosystem functions and conditions can occur as a result of natural and anthropogenic influences, both of which can be the result of

similar stressors such as fire or flood and have similar effect on the receiving environment. Fire, sedimentation and food procurement by animals are natural stressors that, in a system unaltered by unsustainable human influences, would have a regular/cyclical, often seasonal influence on ecosystems (Kent & Coker 1992). These events and/or their combination into a cyclical regime are termed disturbances because they interrupt the successional development of a plant community (*e.g.* Grime 1979, Van der Valk 1992). This is a specifically plant-centric view of disturbance and whilst other definitions of 'disturbance' exist, such as those affecting invertebrates or hydrology, the focus of the present study is specifically on the use of plants to assess the influence of disturbance on wetland ecosystems. Recognizing that humans and other animals have influenced disturbance regimes for a long time, often shaping so called natural environments, sustainable and small-scale disturbances such as veld burning and traditional harvesting regimes are taken to be natural and considered to have low impact (Kent & Coker 1992). A change in these natural disturbance regimes, such as interruption of, or increased intensity or frequency of fire, or flooding, or grazing pressure, as a result of impacts from unsustainable human development or expansion are, however, regarded as unnatural disturbances that cause impairment of the natural environmental condition (*e.g.* Kent & Coker 1992, Deacon 1992, Clarkson *et al.* 2004). This concept of alteration from a natural condition stems from the work of Karr *et al.* (1986) in that the reference condition is considered to be a natural state, unaltered by human interference. The reference condition of full ecosystem integrity is a state in which the ecosystem has ecological and biotic integrity as inferred from the "….*ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of natural habitat …*" (Karr 1991). In this definition the *species composition, diversity and functional organization* are taken to reflect environmental condition, spawning the concept of using the present state of the biota to infer an environmental condition and hence the potential for biological assessment or 'bioassessment'. In most areas some form of large landscape-scale disturbances are natural and an equilibrium state may exist in which patchy disturbance is balanced by regrowth. In others where equilibrium does not exist due to long-lasting disruptive effects, naturalness is more difficult to define and several communities of plants may be natural for any given site at any given time (Sprugel 1991).

In the present study, the impact of anthropogenic influences that cause unnatural impairment of ecosystem or environmental conditions is considered to constitute anthropogenic disturbance and is often referred to, for brevity, as disturbance. An event

or other stimulus and/or a landuse that causes stress or change to an organism or an ecosystem are referred to in this thesis as "stressors". The influence of disturbances unrelated to human activity is not considered to constitute a part of the present research. Recognition of the period of time that has elapsed after a natural/unnatural disturbance event is necessary, however, in order to be able to identify the influence of succession on current community structure (Van der Valk 1992, Middleton 1999) and to recognize the time lag in response to habitat loss and fragmentation (Helm *et al.* 2006).

Anthropogenic disturbances to wetlands are the result of numerous recognized stressors and an extensive review of stressors known to impact wetlands and examples of the response of the biological assemblage are presented by Adamus & Brandt (1990) and Adamus *et al*. (2001). Human landuse, at both localized (within wetland) and large landscape or regional scales (within catchments and or biomes), impact on natural environmental parameters that determine habitat availability for the biota and are therefore considered disturbances that can alter the biotic community within an ecosystem (e.g. Adamus & Brandt 1990, Sala *et al.* 2000, Aznar *et al.* 2003, Declerk *et al.* 2006). Physical alteration of wetland habitats occurs as a result of geomorphological and hydrological changes (i.e. water residence time and flow dynamics: quantity, quality and temporal aspects of flow), alteration of nutrient and mineral availability, sediment and organic loading and resultant turbidity and oxygen availability, direct vegetation alteration by alien species invasion, vegetation utilization and/or land clearance, infilling and utilization for non-natural purposes (Adamus & Brandt 1990, Adamus & Gonyaw 2000, Dahl 2004). Such geophysical and biological impacts totally or partially alter the physical landscape, cause considerable wetland habitat fragmentation, and alter biodiversity (*e.g.* US EPA 2002a, Adamus *et al.* 2001, Fore 2003, Clarkson *et al.* 2004, Rountree *et al.* 2007, Macfarlane *et al.* 2008). Whilst complete changes in physical structure of the landscape are obviously apparent as disturbances that will affect the natural functional integrity and environmental condition of wetlands, there are many less obvious stressors that degrade these ecosystems. For instance, it has been shown that alteration of biotic assemblages often changes the balance of ecosystem functionality and integrity, further altering the environmental condition of the impacted wetland (*e.g.* Keddy 2000, Mitsch & Gosselink 2000, US EPA 2002a). The effects of biotic exchange (deliberate or accidental introduction of plants and animals to an ecosystem) constitute a negative effect on the biodiversity of many areas and have particularly noticeable impact on systems that are isolated, such as mediterranean climatic regions and freshwater ecosystems (Sala *et al.* 2000). The impact of introduced alien species is often to replace indigenous species and

thereby to homogenize the local collection of species with that of the global collection (Slobodkin 2001. In the likely event of incomplete homogenization due to biotic exchange, all portions of invaded ecosystems will initially increase in diversity, but portions with high native diversity will not increase as much as portions with low native diversity (Collins *et al.* 2002). Whether competition from introduced species threatens indigenous biodiversity is questioned by some as introduced plant species have seldom caused extirpations through competitive exclusion (Davis 2003). The impact of invaders on the cover, abundance and distribution of native species and on the functioning of ecosystems is, however, of greater importance than their effects on species diversity (Collins *et al.* 2002).

Climate change is another anthropogenic source of stress that is likely to alter the distribution and abundance of species and thereby change community assemblage patterns (Midgley *et al.* 2002). Whilst some species will be able to track favourable climate envelopes, others will lag behind, resulting in range dislocation (i.e. no overlap between current range and future predicted potential range) (Midgley *et al.* 2002). Species most at risk from the impacts of climate change are those with narrow habitat ranges. As isolated wetlands are islands of habitat within a terrestrial sea, wetland plants and their habitat have relatively fragmented geographical distribution and are thus susceptible to species losses as a result of range dislocation. Climate change is likely to have a similar impact on all wetlands within a region with similar climatic constraints. The disturbance potential that climate change poses may exacerbate other stresses on the receiving environment. During the last two million years – the Pleistocene – cool and dry glacial climates predominated within the Fynbos Biome rather than the warmer periods of interglacial climate such as at present. The maximum depression of mean annual temperatures during glacial maxima were in the order of 5^0C lower than present temperatures; with mean summer temperatures being close to present mean winter temperatures. Pleistocene glacial climates were drier in general than at present because evaporation from cooler oceans was lower, but the winter dominated season of precipitation would have existed much as at present (Deacon *et al.* 1992). A loss of Fynbos biome area of between 51% and 65% is projected by 2050 (depending on the climate scenario used). Due to thermal and drought stress the majority of this areal loss is predicted to occur in northerly (equatorward) latitudes, distributed more or less evenly with changing altitude. Increasing minimum temperatures along the western seaboard may also reduce the range of the Fynbos Biome in this area (Midgley *et al.* 2002). Assessment and monitoring for the impacts of climate change would therefore require

baseline information on the present ecological state of wetlands and their biotic assemblages in order to interpret the future impacts of climatic change. The present study provides much useful baseline information but does not further attempt to assess or monitor the impacts of climate change.

Anthropogenic disturbances that have already impacted the Fynbos Biome, and are ongoing despite considerable conservation effort, are from land transformation for agriculture and urban development, and transformation of the natural phytosociology due to invasive alien vegetation and associated biotic exchange and disruption to ecosystem processes (Rouget *et al.* 2003). The disruption of ecosystem processes by biotic exchange is considerable in at least three arenas. The impact of alien vegetation on fire regimes has had a substantially negative impact on the phytosociology of indigenous vegetation in the Fynbos biome as in other mediterranean climates (Van Wilgen & Richardson 1985, Van Wilgen et al. 1990, Keeley et al. 2005). The Leaf litter from Australian acacias significantly increases the nutrient content in the rooting zone of Fynbos vegetation (Witkowski & Mitchell 1987), such alien vegetation can itself thus be considered a negative disturbance in low nutrient environments such as Fynbos. And the considerably higher biomass of dense stands of invasive tree species (e.g.: Eucalypts, Poplars and Acacias) results in higher water use than indigenous vegetation (Le Maitre *et al.* 2002) and thus also constitutes a negative impact. Invasive alien plants thus constitute a considerably negative influence in the Fynbos biome. These numerous individual stressors do not necessarily impact on an ecosystem in the same way or result in the same response by the biotic community. Within South Africa, very limited information is known about the response by the wetland biotic community to the impacts of these stressors (Macfarlane *et al*. 2008).

1.2.2 Plants as indicators of environmental condition

Plant communities are fundamental biological components of many wetlands, and as such, to a large extent they define the biological and ecosystem characteristics of these wetlands (*e.g.* Mitsch & Gosselink 2000, Keddy 2000). Plant species typically have a range of environmental conditions in which they are able to exist and changes to any of these can change the community assemblage of plants in a habitat (*e.g.* Whittaker 1962, Grillas 1990, Bonis *et al.* 1995, Rebelo *et al.* 2006). The presence of certain plant species and their resultant assemblage in a community therefore represents a particular set of environmental determinants that exist at a given location. Macrophytes vary greatly in their anatomy, physiology, life-history traits, and ability to tolerate inorganic and

biological stressors and as such they may be indicative of a variety of ecosystem stressors including nutrient runoff and changes in hydrologic regime (Lacoul & Freedman 2006). Some macrophytes have a cosmopolitan distribution and display high levels of polymorphism and phenotypic plasticity in response to variations of environmental factors; these qualities allow them to occur over a wide range of conditions. Other species, however, have narrower tolerances and are potentially useful indicators of environmental conditions, in terms of either their presence or relative abundance within communities. Whilst certain plants have been shown to indicate the presence of chemical constituents (Lukács *et al.* 2009) metals (Balonson & Mal 2005) or water quality (Lougheed *et al.* 2001) their presence alone does not guarantee the existence of certain environmental conditions. The community assemblage, representative of a number of species, provides more information about the set of environmental determinants at a given location than can be determined from single species (Magurran 1992). The development of vegetation as bioindicators has been most successfully achieved as based on the community composition and vegetation attributes such as species richness and density rather than the use of single species (Fore 2003, Dahl 2004, Balonson & Mal 2005).

Wetland plant communities are products of biological interactions (competition, facilitation and herbivory) and causal environmental factors including nutrient availability, hydrological conditions, sediment fluxes and fire as well as human influences on these factors (*e.g.* Keddy 2000, Mitsch & Gosselink 2007). The hydrology and geomorphology of a landscape are (hydrogeomorphic) drivers of wetland types (Brinson 1993), and are potentially important determinants of habitat availability for plants (*e.g.* EUR15 1999, Mucina *et al.* 2006a). Along with other natural environmental determinants of biotic distribution such as climate and geology these parameters partition a landscape into areas of different habitat that will support different communities of plants (*sensu* Walter 1973, Cowan 1995). Under natural conditions, unaltered by human disturbances that degrade or impair the integrity of a habitat, characteristic sets of species, or communities of plants, can be expected to exist within similar habitat (Whittaker 1953). The comparison of the vegetation of natural (reference) and impaired environments has proven useful in the identification of indicator species or vegetation attributes that are considered characteristic of these different conditions (Section 1.2.3).

Attributes of biotic communities that furnish a dependable and repeatable assessment of a wetlands environmental condition are considered as potentially useful bioindicators

(Karr & Chu 1999). An ideal metric has a constant "quantitative" difference over a range of anthropogenic disturbances at different intensities (Karr & Chu 1999). For instance species richness of native perennials was shown to give a notable change over a disturbance gradient that was constructed with a number of different disturbances occurring in and around seasonal wetland plant communities (DeKeyser *et al.* 2003).

In comparison to short-lived species such as fish, invertebrates and diatoms, the relatively long life span of vegetation means that vegetation has considerable inertia to changes that are not immediately deleterious such as massive flood events. The inertia that plants display to changes in ambient environmental conditions is an important consideration when attempting to relate species composition, as a representation of present ecological state, to present environmental conditions (Helm *et al.* 2006). The inertia of vegetation response to environmental change means that there are lag effects in terms of alteration to species composition and dominance (Von Holle *et al.* 2003), and possible local extirpation events that are waiting to occur as a result of habitat loss and fragmentation (Helm *et al.* 2006). Thus whilst the existing community of vegetation may correlate with levels of human impact thus showing an expected relationship and suggesting just cause for the acceptance that species or vegetation attributes should be considered as bioindicators (Fore 2003), such metrics also need to be reversible. If the human stressors are reduced the indicators should reflect that change and vegetation inertia means that this is often not the case, and that the vegetation reflects a perturbation event at some time in the past. Little appears to have been published about this issue in the literature of phytoassessment of wetlands but the combination of numerous metrics in an index of biological integrity is considered as being likely to give an accurate result of present environmental state (US EPA 2002a).

The issue of vegetation inertia is particularly problematic with relation to development of bioindicators using alien invasive species. Many of invasive plants are early seral species that establish in low-competition environments created by disturbance events (Planty-Tabacchi *et al*. 1996) yet once established may persist for long periods after the disturbance event that facilitated their establishment (Richardson *et al.* 2007). The debate about whether invasive species are the drivers or passengers of change in degraded ecosystems is considerable (MacDougall & Turkington 2005), however, the presence of considerable amounts of alien vegetation suggests a negative impact on the wetland ecosystem. The use of the diversity, cover or dominance of alien vegetation as

metrics for phytoassessment is therefore considered to have considerable value despite the potential complication caused by vegetation inertia.

1.2.3 Phytoassessment

Bioassessment is based on the premise that the distribution and assemblage patterns of the biota are determined by both the underlying natural environmental template and by superimposed anthropogenic disturbances (Karr 1981). Measurement of the difference in patterns of species assemblage in reference relative to disturbed wetland habitat of the same type can be used to assess wetland ecosystem condition (*e.g.* Karr 1981, Reynoldson *et al.* 1997). Organisms such as plants can therefore be used to represent the environmental determinants and a state of environmental condition that exists in a given wetland. As such, predominantly in the USA, macrophytes have been successfully used as an indication of wetland environmental condition relative to levels of anthropogenic disturbance or land-use (*e.g.* Fennessy *et al.* 1998, Galatowitsch *et al.* 2000, Mack 2001, Simon *et al.* 2001, Lopez & Fennessy 2002, Gernes & Helgen 2002, Fennessy *et al.* 2004, Mack 2007). International approaches to assessment using plants (phytoassessment) compare wetlands with natural similarity as determined by ambient environmental parameters, but different degrees of anthropogenic landuse in order to identify vegetation attributes with characteristic affinity for minimally impaired *vs* impaired conditions. This avoids the problem, as pointed out by Sprugel (1991), of trying to define what constitutes natural or "Reference" communities from the perspective of the vegetation, relying rather on the degree of anthropogenic disturbance or landuse as a measure of impairment.

1.2.4 Wetland bioassessment research in South Africa

Although methods of bioassessment have been developed for riverine ecosystems in South Africa (Dickens & Graham 2002, Kleynhans *et al.* 2007), there has been very little development of methods for bioassessment of wetland ecosystems. In other countries diatoms, invertebrates, algae, vascular plants, fish, birds and reptiles have all been used with varying degrees of success for bioassessment of wetland environmental condition (*e.g.* Adamus *et al.* 2001, US EPA 2002a, Butcher 2003, Fore 2003). Whilst no vegetation-based assessment protocols have been developed specifically for wetlands in South Africa, methods of measuring the cumulative amount of anthropogenic impact on environmental conditions (Macfarlane *et al*. 2008) and habitat integrity (Rountree *et al.* 2007) have included qualitative assessment of some aspects of the vegetation. WET-

Health incorporated disturbance to vegetation and departure from an anticipated reference condition of the plant community as an aspect of assessment of wetland condition (Macfarlane *et al.* 2008). The Wetland Index of Habitat Integrity (WIHI), measures present ecological state by assessing the intensity and extent of landuse with regard to its anticipated alteration of the natural vegetation cover (Rountree *et al.* 2007). WET-Health and WIHI do not, however, claim to determine the condition of the biological community and their development did not incorporate any empirical comparison of the vegetation of wetlands with different degrees of disturbance. The intent of the present research was twofold to examine the data set collated for the Water Research Commission project (Corry in press) to:

- Identify geographical regions of the coastal lowlands of the Fynbos biome with homogenous wetland vegetation; and within which to
- Perform an empirical comparison of impaired *vs.* minimally impaired (reference) wetland plant communities in order to identify attributes of the vegetation assemblage that can inform assessment of environmental condition.

1.3 State of wetland plant ecology in South Africa

In South Africa due to the past perception of wetlands as being waste spaces, for which agricultural subsidies existed to facilitate their conversion into agriculturally productive land, limited research has been carried out in understanding the variability and biodiversity that they encompass (*e.g.* DWAF 2004, Dallas *et al.* 2006, Mucina *et al.* 2006a). Uys (DWAF 2004) and Dallas *et al.* (2006) highlight the lack of wetland ecological research in South Africa and Cook (2004) and Mucina *et al.* (2006a) specifically point out the dearth of information available on wetland plant ecology and plead for a better understanding of species distribution, diversity and abundances and their determinants. It was recognized that the distribution of wetlands and their biotic and abiotic character both reflects and modifies their physical surroundings (Silberbauer & King 1991a & b, King & Day unpublished data). In a review of wetland types, wetland biota, threats to wetlands and to their biota, and of the conservation status of wetlands in South Africa (Cowan (ed.) 1995), Cowan (1995) divided South Africa into regions considered to hold different wetlands based on the interaction of geomorphology and climate (See Section 1.4.4.1 & Figure 1.1). These wetland regions developed by Cowan (1995) have not been tested in order to identify whether each holds wetlands with similar habitat for organisms and could thus be considered different biogeographical regions.

Separation of different types of wetlands in South Africa initially focused on the hydrogeomorphic types of Brinson (1993) and Dini *et al*. (1998). Later classification attempts incorporated further discriminators in order to be able to separate wetlands for remote-sensing-based mapping purposes (Ewart-Smith *et al.* 2006), and for discriminating different habitats for biotic assemblages (SANBI 2009). Although a number of the discriminators employed in the National Wetland Classification System have proven useful for classification of different habitat in Europe (Eur15 1999, Rodwell *et al.* 2002), empirical testing of whether these discriminators separate different wetland vegetation habitat has not been performed in the South African context. What constitutes naturally similar wetland vegetation habitat (i.e. that would hold the same species given similar environmental conditions) is therefore not well understood in the South African context.

1.4 Biogeography and classification of similar spatial units of vegetation

1.4.1 Biomes

Typically plants have tolerance ranges of various physical and chemical conditions within which they can survive and along a gradient of change of any of these environmental parameters, the assemblage of the community structure can therefore be expected to change (*e.g.* Walter 1973, Omernik 1987, Olson *et al.* 2001, Mucina *et al.* 2006a). A change in community structure or assemblage is constituted by a change in the identity, relative abundance and or cover of individual species within said community (Anderson *et al.* 2011). In many instances these gradients that are driving biotic differences are a consequence of geography and geology as the climate, soil nutrients and soil physical properties determine the growing medium in a given habitat. Hence in general, plant communities from similar habitats within a biogeographical region can be expected to be more similar to each other than communities from similar habitats but different regions. On this basis and as a first step in classifying vegetation into broadly typical units, Walter (1973) split land masses of the world into areas (zones) of distinct climatic and ecophysiological conditions in what he called zonobiomes. Walter (1973) presented an essentially ecological concept, considering biome as an area of uniform environment, belonging to a zonobiome, which is defined by the climatic zone where it is found (Coutinho 2006). The boundaries of each zonobiome are determined from seasonality of temperature and precipitation and correlate with conditions of moisture and cold stress that are strong determinants of plant form, and therefore the vegetation that defines the

region. The Mediterranean zonobiome, where ever it occurs around the globe, is characterized by winter rain and summer drought with sclerophyllous vegetation (i.e. with drought-adapted leaves), particularly dominated by frost-sensitive shrublands and woodlands. The Fynbos biome (Rutherford *et al.* 2006) is one of the nine biomes that exist in South Africa; and is a representative of the global Mediterranean zonobiome (Walter 1973, Walter & Box 1976).

1.4.2 Broad terrestrial vegetation units within the Fynbos biome

The Fynbos biome contains three quite different, naturally fragmented terrestrial vegetation units, Fynbos, Strandveld and Renosterveld, which are further separated into 12 bioregions (see below). The transition between Fynbos and Renosterveld is predominantly dependent on differences in leaching as determined by annual precipitation with more leached and oligotrophic soils supporting Fynbos (Cowling & Holmes 1992). The boundary between Fynbos and Strandveld is proposed by Rebelo *et al.* (2006) to be largely determined by fire dynamics with Sand-Fynbos occurring adjacent to Strandveld which, has higher succulent coverage (and thus a lower fuel load) supported by nutrients derived from salt spray from the sea. Renosterveld and Strandveld occur on different soil types, predominantly shale and sand respectively, and typically the sand/shale interface is with acid sands supporting Sand-Fynbos rather than Strandveld such that Renosterveld is typically surrounded by Fynbos (Rebelo *et al.* 2006).

1.4.3 Bioregions & terrestrial vegetation units

Based on the ecoregions concept of Olson *et al.* (2001), which focused on terrestrial regions of the world, as defined by distinct assemblages of species, Rutherford *et al.* (2006) created 35 vegetation bioregions for South Africa, focused on the floristic composition of the component dryland (terrestrial) vegetation units. The terrestrial vegetation units already grouped by association with a geological substrate were thus further grouped by climatic similarity. In areas with a complex mosaic of vegetation units these bioregions group units of vegetation that are spatially disparate, being geographically dispersed and separated by other bioregions. The close proximity and intermingling of very different terrestrial vegetation units in the Fynbos Biome make it difficult to establish bioregions with strictly distinct floras and the 12 bioregions in this biome include, in some cases, a combination of Fynbos and Renosterveld and were based instead upon the differences in climatic conditions (Rutherford *et al.* 2006). These

bioregions are sub-ordinate spatial units to a biome (Rutherford *et al.* 2006) and differ from the ecoregions of Kleynhans *et al.* (2005) and the bioregions of Brown *et al.* (1996) in that different criteria are used to differentiate regions in each case. In the Fynbos Biome, therefore, these bioregions of Rutherford *et al.* (2006) describe areas of broad macroclimatic and geological similarity.

Within the Fynbos Biome, at a finer spatial scale than the bioregions, the edaphic (soil type and nutrient) dependence of many spatially smaller units of terrestrial vegetation mean that geological substrate type and geomorphic land units were used as surrogates for determining the boundaries of distinct terrestrial vegetation units (Rebelo *et al.* 2006). For instance the Cape Flats Sand Fynbos that occurs within acidic sands in planar landscapes can be contrasted with Cape Flats Dune Strandveld that occurs within calcareous and alkaline dune sands.

Another means of grouping wetlands within a regional context as based on similar geological and climatic conditions has been used to create the National Freshwater Ecosystem Priority Areas (NFEPA) (Roux *et al.* 2006). The NFEPA wetland vegetation groups were derived from grouping the 438 terrestrial vegetation types (*sensu* Mucina & Rutherford 2006) into 133 wetland vegetation groups with similar geological and climatic conditions.

1.4.4 Wetlands as anomalous habitat units within the zonobiome concept

Increased water availability in wetlands creates ephemerally to permanently waterlogged or inundated land as an anomalous habitat surrounded by an unsaturated terrestrial habitat. Wetlands are considered to represent anomalous environments within the complex of the macroclimatically-determined zonobiome scheme (Walter 1973) because hydrogeological conditions and/or saline substrates (soil types or bedrock) create habitats that are atypical of the climatic zone (Walter & Box 1976, Walter 1985, Mucina *et al.* 2006a). In contrast to the zonobiomes and resultant so-called zonal vegetation, azonal vegetation types are those influenced to a greater extent by factors other than the macroclimatic constraints of temperature and moisture availability that are used to define the zonobiomes (Walter 1973, 1985). Many wetland species were reported to have broad to (sub)cosmopolitan distribution that was not constrained by climatic determinants of temperature and moisture availability and, as they occur across many zones or zonobiomes, wetland vegetation was thus considered azonal (Walter 1973). Santamaría (2002) argues that many wetland taxa have broad distribution across multiple zonobiomes as a result of successful dispersion events and that climatic factors have

limited effects on wetland plant species distribution. Somewhat contradictorily, however, Santamaría (2002) also recognized, that climatic constraints on wetland species distribution resulted in major global scale zonobiome disjunctions such as between the mediterranean, tropical, temperate and subarctic flora, therefore suggesting that wetland species distribution is in fact zonal. This zonal distribution of wetland vegetation was backed up in part by the findings of Peinado *et al.* (2007), which showed broad climatically determined phytogeographical regions (phytochoria) or zonobiomes (*sensu* Walter 1973, 1985) each with characteristic wetland vegetation along the Pacific coast of North America; however, these researchers also suggested that wetland vegetation was azonal as a result of soil types that exerted a greater influence than climate on the distribution of wetland vegetation. Peidno *et al.* (2007) also showed that the latitudinal distribution limits of the azonal vegetation practically coincided with those already established for the zonal vegetation. It is perhaps apparent, therefore, that there is no clarity on whether wetland vegetation should be considered zonal or azonal in terms of the macroclimatic drivers of distribution. Whilst there are many wetland species with distribution across multiple zonobiomes, their distribution within zonobiomes is determined by the same edaphic drivers of the distribution of the terrestrial vegetation within zonobiomes.

In wetlands in South Africa the concentration of salts and/or the hydrological regime created by levels of waterlogging, flooding and tidal influence are considered to exert an influence greater than the macroclimate on floristic composition, structure and dynamics resulting in broad and azonal distribution of wetland plants (Mucina *et al.* 2006a). In the most recent mapping of wetland vegetation in southern Africa (Mucina *et al.* 2006a), where a wetland vegetation unit occurs exclusively within a climatic zone (biome) then it was considered to be *intrazonal* (i.e. occurring exclusively within a zone or zonobiome) and where it occurs irrespective of climatic and vegetation zones it was considered azonal.

Wetland vegetation units in South Africa are predominantly considered distinct from the surrounding zonal (or terrestrial) vegetation units except in the Fynbos freshwater marshes and seeps dominated by endemic *Capensis* elements that are embedded within the shrublands of the Fynbos Biome (Mucina *et al.* 2006a). In most other wetland types within the Fynbos Biome Mucina *et al.* (2006a) considered wetland taxa as not necessarily Fynbos, Strandveld or Renosterveld affiliates and consequently anticipated

that wetland species would have widespread distribution that spanned across these terrestrial vegetation units.

In order to map the spatial distribution of distinct units of wetland vegetation, Mucina *et al.* (2006a) classified wetland vegetation into distinct ecological and phytogeographical units. The anomalous nature of the wetland vegetation in the zonobiome scheme meant that it needed to be classified independently of the zonal (terrestrial) vegetation. This approach to classifying wetland vegetation units followed multi-layered criteria, with hydrodynamics and/or salt content as azonality-driving macro-ecological factors and further divisions along the broad phytogeographical boundaries of the biomes (Mucina *et al.* 2006a). The classification of different types of inland wetlands was achieved by separation according to these macro-ecological criteria. The vegetation of "freshwater wetlands" with stagnant or slow-flowing waters (lentic) differs from "alluvial vegetation" that fringes water courses characterized by flowing water (lotic conditions) and undergoing dynamic change due to a periodic flood regime (Mucina *et al.* 2006a). The addition of allochthonous nutrients resulting from water flowing into or through a riverine wetland (Wetzel 1975), and their periodic scouring in years of high flow (Keeley & Zedler 1996), emphasize differences between the Riparian and Alluvial vegetation units (AZa1 & 2) of riverine wetlands in the Fynbos Biome relative to the isolated, autochthonous and often nutrient poor Vernal Pool (AZf2) and Cape Lowland Freshwater (AZf1) vegetation units of Mucina *et al.* (2006a). A third vegetation class of inland wetland vegetation, "inland saline vegetation" comprises vegetation accompanying salt-laden intermittent rivers and salt-pans in which salt is the major ecological determinant. Endorheic depressions in saline substrates, or wetlands fed by saline water sources, result in higher concentrations of both water and salt than the surrounding terrestrial uplands. Further division within these classes, at the scale of biomes, follows biogeographical (floristic) criteria reflecting correlation with abiotic determinants that are also driving the terrestrial matrix of zonal vegetation of surrounding zonobiomes (Mucina *et al.* 2006a). For the freshwater inland lentic subset of the above wetlands, the subdivisions of vegetation based on association to zonobiomes for South Africa are presented in Table 1.1. For tables of the lotic and saline inland wetlands see Mucina *et al.* (2006a).

Table 1.1: Spatial links between the freshwater inland lentic wetland vegetation units and the surrounding biome, with reference to the zonality status of the units and the broad landscape within which they are found (after Mucina *et al.* 2006a).

Atlas Code	Freshwater Wetland Vegetation Units	Biome	Zonality	Broad Wetland Landscapes (Cowan 1995)

AZf 1	Cape Lowland Freshwater Wetlands	Fynbos	intrazonal	S: Coastal Slope
AZf 2	Cape Vernal Pools*			
AZf 3	Eastern Temperate Freshwater Wetlands	Grassland	intrazonal	P: Plateau
AZf 4	Drakensberg Wetlands			M: Mountain
AZf 5	Lesotho Mires			
unmapped	Cape Mountain Wetlands	Fynbos	intrazonal	
AZf 6	Subtropical Freshwater Wetlands	Savannah; Albany Thicket; Indian Ocean Coastal Belt	azonal	C: Coastal Plain + S: Coastal Slope + P: Plateau

*Vernal pools, or the context in which this term is used in the present study, will be described below

Cowan's (1995) terminology is somewhat confusing as only the Pongola/Mkuze region (Ce in Figure 1.1) was considered to be coastal plain whilst the areas such as Agulhas Plain and the Cape Flats, which fall within the Cape Forelands (coastal lowlands), were considered to constitute coastal slope. The Cape coastal lowlands are, however, generally described as having planar topography (Lambrechts 1979, Kleynhans *et al.* 2005, Schulze 2006) and wetlands dominated by Cape Lowland Freshwater vegetation may be more accurately considered as occurring in both coastal plain and coastal slope landscapes.

1.4.4.1 Wetland regions

At a broad landscape scale Mucina *et al.* (2006a) classified the wetland vegetation units with "high certainty" into the four broad geomorphological categories of Cowan's (1995) wetland regions, with the notable exception of the azonal subtropical units which are shared by three of the four landscapes, not occurring in the mountain wetlands. Cowan (1995) subdivided the four broad landscape categories, using geomorphological and climatic characteristics, including temperature and moisture balance, into 26 regions as depicted in Figure 1.1 (for the full list of these regions see Cowan 1995). No attempt was made by Mucina *et al.* (2006a) to further subdivide the wetland vegetation units according to these 26 wetland regions. The inset map in Figure 1.1 depicts the full extent of the mediterranean region of the Western Coastal Slope (SW.m) which was the main focus of the present study. Along with the temperate region of the Southern Coastal Slope (SS.a) and the desert region of the Western Coastal Slope (SW.w) these three regions constitute the full extent of the Cape coastal lowlands. The 26 regions contain wetlands with similar topography, hydrology and nutrient regime (as determined by temperature and moisture gradients as well as by geological stratigraphy) and were thus considered

likely to contain similar sets of biota (Cowan 1995). Cowan (1995) anticipated that within the 26 regions, local differences in geology would determine minor vegetation types as determined by local nutrient availability. Influence of local nutrient differences would, however, suggest that the wetland vegetation may not in fact be truly azonal and that geological substrates that drive zonal or zonobiome differences (*sensu* Walter 1973, Rutherford *et al.* 2006) may also drive wetland vegetation unit differences. This concept of local edaphic influences (Cowan 1995) is not entirely compatible with the intrazonal wetland vegetation units of Mucina *et al.* (2006a), which are relatively much larger spatial units thought to contain uniform wetland vegetation.

Within the geographical area of the Fynbos Biome, Cowan's wetland regions are typically broader than, and often incorporate, several of the zonal bioregions (*sensu* Rutherford *et al.* 2006) and multiple terrestrial vegetation units (Rebelo *et al.* 2006). Within the Cape coastal lowlands, the units of wetland vegetation mapped by Mucina *et al.* (2006a) are spatially broader than the wetland regions of Cowan (1995). For instance "Cape Lowland Freshwater", "Cape Vernal Pools" and "Cape Lowland Alluvial" freshwater wetland vegetation units extend across both Cowan's "mediterranean Western Coastal Slope" region (SW.m) and "temperate Southern Coast Slope" region (SS.a) (See Figure 1.1). If wetland vegetation is truly azonal and driven by the macroclimatic influences of hydrology and salinity then it would be correct to suggest that the spatially broad vegetation units classified by Mucina *et al.* (2006a) would have some uniformity across the relatively diminutive wetland regions of Cowan (1995). If, however, local geological differences (and potentially also climatic differences) do affect the phytogeography of wetland vegetation, then not only would there be different classes of wetland vegetation in different Cowan wetland regions, but potentially sub-classes within those regions as determined by local substrate differences. Such small scale differences are apparent between units of terrestrial vegetation in the Fynbos biome (*sensu* Rebelo *et al.* 2006) and their potential influence on wetland vegetation is acknowledged by Mucina *et al.* (2006a).

1.4.4.2 Wetland vegetation units

The classification and mapping of different units of wetland vegetation by Mucina *et al*. (2006a) was based on a meta-analysis of relatively sparse data gleaned from temporally and spatially widely dispersed studies. Whilst many studies have included the vegetation of inland wetlands (for lists see Thompson *et al*. 1985, Rogers 1997, Mucina *et al*. 2006a) most focused on small areas or gave wetland vegetation only marginal attention whilst

focusing on the terrestrial communities (Mucina *et al*. 2006a). The distribution of wetland vegetation species and the floristic composition of different wetland units is not well documented in South Africa (Cook 2004, Mucina *et al.* 2006a), and data such as that collated in the present study can inform our understanding of phytogeography. The successional development, not only of wetland vegetation (Middleton 1999) but of the landforms (hydrogeomorphic units) in which they are manifest (Breen pers. com., Tooth & McCarthy 2007), and the annual seasonality of water level (hydroperiod) and nutrient load (Mitchell & Rogers 1985) means that comparison of data sets from different time periods, and even from different seasons within a year, may not reflect similar vegetation for the same wetland. Mucina *et al*. (2006a) recognized the need for more rigorous data treatment in order to accurately classify different units of wetland vegetation. They considered their mapped wetland vegetation units as being at a coarse scale and that further division might be possible along biogeographical lines and corresponding with the abiotic determinants (climate and soils) that drive the divisions in the terrestrial matrix of zonal vegetation (*sensu* Rebelo *et al.* 2006). What constitutes phytogeographical units of comparable freshwater habitat sufficient for the purposes of phytoassessment development, i.e. with limited internal natural variability, was thus unclear from available material at the start of the present study. It was therefore hypothesized that wetlands from the Cape Lowland Freshwater vegetation unit (AZf1) (Mucina *et al.* 2006a) as contained within the mediterranean Western Coastal Slope region (SW.m) (Cowan 1995) on the Cape coastal lowlands would have sufficient community similarities within which wetlands impaired by landuse would have identifiably different vegetation from minimally impaired wetlands. Such an eventuality would facilitate the development of phytoassessment tools that could be used to assess environmental condition of wetlands in this region.

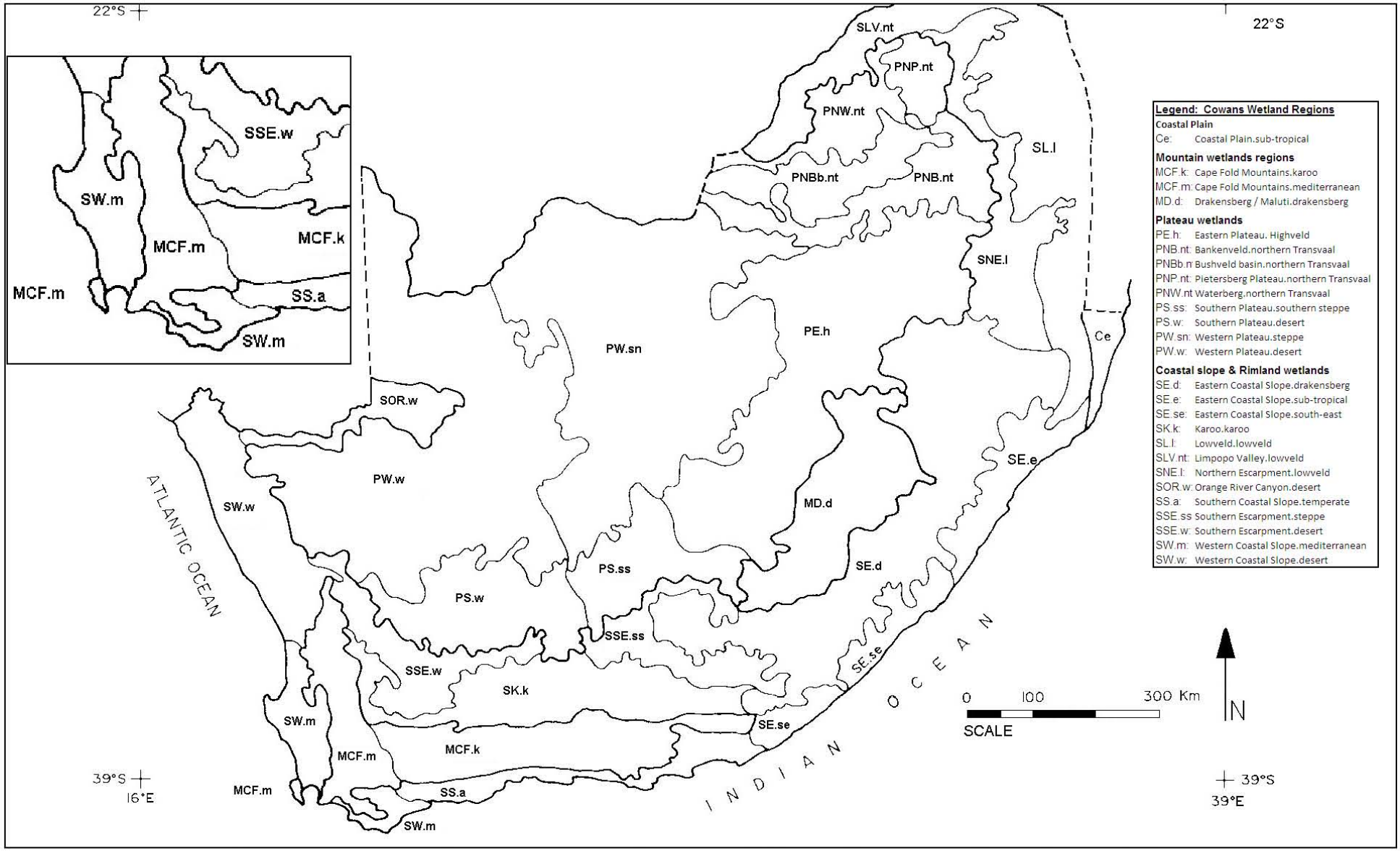


Figure 1.1: Cowan's Wetland Regions of South Africa (After Cowan 1995). Inset showing the totality of the South-western mediterranean (SW.m) coastal slope region which was the area of focus for the present study.

i. Cape Lowland Freshwater wetland vegetation

The Cape Freshwater Lowland (AZf1) vegetation unit is recorded and mapped as occurring in association with flat and depression landscapes including the fringes of large bodies of water such as coastal lakes from Verlorenvlei on the Cape west coast to the Wilderness Lakes on the Cape south coast (Mucina *et al.* 2006a). These wetlands are embedded within various terrestrial zonal vegetation units of Strandveld, Renosterveld and Fynbos of the Fynbos Biome (Mucina *et al.* 2006a).

ii. Cape Vernal Pool vegetation

Vernal pools in the Fynbos Biome are described by Mucina *et al.* (2006a) as seasonal or ephemerally waterlogged wetlands, usually inundated by between 2 to 10 cm at the height of the wet season (in Winter and onward into Spring) and support a vegetation unit other than the emergent Cape Lowland Freshwater vegetation unit that is more typically associated with inland freshwater wetlands with permanent waterlogging and seasonal to permanent inundation. In the arid south-western Cape, vernal pools become apparent after the onset of winter rain has saturated the soil pore space and water begins to pool above ground on the soil surface. This is common to the definition of vernal pools in most mediterranean regions of the world as described by Keeley & Zedler (1996). Their definition of vernal pools, as *precipitation-filled seasonal wetlands inundated during periods when temperature is sufficient for plant growth, followed by a brief waterlogged-terrestrial stage and culminating in extreme desiccating soil conditions of extended duration,* was built on the classification by Cowardin *et al,* (1979) of vernal pools as seasonally flooded emergent wetlands of the palustrine system. Whilst Keeley & Zedlers definition does not explicitly exclude seasonal pools filled by snow-melt, vernal pools in the lowlands of the mediterranean regions of the world, and particularly in the Fynbos Biome, are typically filled by rain-water. A factor that differentiates Vernal Pools from seasonal pools, as well as permanent wetlands, is the lack of water input by long distance drainage. Therefore, unlike formal classification systems such as the Cowardin Scheme, Keeley & Zedlers Vernal pools do not include stream-fed or allochthonous seasonal wetlands. These definitions suggest similarity of the vernal pool habitat to the seasonally saturated zone surrounding isolated-wetlands that experience seasonal hydrological fluctuation (See Figure 1.2).

The difference in vegetation found in Cape Vernal Pools relative to the seasonally saturated zone of Cape Lowland Freshwater wetlands may therefore not be significant and so called Vernal Pool endemics may simply be a construct of insufficient baseline sampling in the ephemerally saturated zone around endorheic but isolated Cape Lowland Freshwater dominated wetlands. For instance, the tenagophytes are plants with their juvenile stage submerged and adults emergent or terrestrial. Whilst many tenagophytic taxa are present in vernal pools they are also found in the temporary waters surrounding permanently inundated zones of freshwater wetlands (Cook 2004). Indeed this begs the question as to what difference in microclimate actually exists in the different hydrogeomorphological units and hydrological zones of wetlands.

1.4.5 Diversity within the Cape Lowland Freshwater vegetation

Within the terrestrial vegetation of the Fynbos Biome there is a high level of diversity (gamma diversity within the whole biome), with a considerable beta diversity or degree of change or turnover along specific gradients, namely the change in the relative cover/abundance or identity of species creating changes in species composition or structure of vegetation communities:

- between the different terrestrial vegetation units along habitat or environmental gradients within local landscapes (such as between different soil types or, within a wetland, between hydrological zones); and
- between the terrestrial vegetation of equivalent habitats separated by different geographical distances.

Both of these forms of variation in community composition are the result of beta diversity *sensu* Whittaker (1960 & 1972) but the change along geographical gradients was also confusingly called gamma diversity by Cody (1975 & 1983).

The high gamma diversity of the terrestrial vegetation in the Fynbos Biome is thought to be due to the extreme amount of change or turnover of macroclimatic factors and soils within the Biome that, in combination with high speciation and low migration rates (Latimer et al 2005), results in high levels of species turnover along these environmental coenoclines (*e.g.* Lambrechts 1979, Cowling *et al.* 1992, Cowling *et al.* 1996, Rebelo *et al.* 2006).

A second recognised aspect of beta diversity is that of variation in community structure among a set of sample units within a given spatial extent (biome, region, unit of vegetation) or within a given category of a factor (such as a habitat or an experimental treatment, or an observational survey) (Anderson *et al.* 2011). Such variation in diversity has direct correspondence with multivariate dispersion or *variance* in community structure (Anderson *et al.* 2006)

Despite the fact that wetland vegetation is considered azonal (Walter 1973, Mucina *et al.* 2006a), the consideration that Cowan's (1995) Western Coastal Slopes region will hold a uniform set of Cape Lowland Freshwater wetland vegetation appears to contradict the acknowledged beta diversity change or extreme turnover in the zonal terrestrial vegetation of the Fynbos Biome along geographical and related climatic and geological coenoclines (Cowling *et al.* 1992). The present study serves as a potential source of information about both the macrophyte turnover between wetlands across the mediterranean Western Coastal Slopes region of the Cape coastal lowlands and across the associated environmental coenoclines between three of its sub-regions (West Coast [western], Cape Flats [central] and Overberg [eastern]).

If the Cape Lowland Freshwater vegetation is divisible into phytogeographically distinct units (with high levels of beta diversity difference between them), then it may be too broad a vegetation unit within which to attempt to develop metrics for phytoassessment, necessitating metric development within each distinct phytogeographic unit.

1.5 Comparable habitat units

A primary step in the development of successful phytoassessment indexes is the identification of comparable groups of wetlands or habitats within them (Mack 2001, Simon *et al.* 2001, Gernes & Helgen 2002, US EPA 2002a). This is done in order that any variation in plant communities that occurs along a gradient of anthropogenic disturbance is not confused with that naturally occurring (1) between regions as a result of differences of macroclimatic and physiogeographic factors or (2) between different habitats created by the different hydrogeomorphology of different wetland types and (3) different habitats created by hydrological zones within wetlands. Major determinants of natural variability between comparable units are thus controlled for; making the detection of differences in ecosystem condition between wetlands more apparent and the

determination of indicator attributes easier (US EPA 2002b). The separation of wetlands into comparable units is done by the classification of phytogeographical regions, wetlands and habitats into classes or categories. Classification is a way of accounting for the effects of natural environmental influences on wetlands and helps avoid comparing wetlands or habitat units of unlike classes. Excessive emphasis on classification, or inappropriate classification, can however impede development of cost-effective and sensible assessment and monitoring programs (Fore 2003). While using too few classes fails to recognize important distinctions among ecoregions (phytogeographical regions), wetlands or habitats and can produce insensitive biocriteria (metrics), using too many adds unnecessary costs to the development of metrics (US EPA 2002c). For the purposes of developing methods for phytoassessment the challenge is to create a classification of ecosystems with only as many classes as needed to represent the range of relevant biological variation in a region and yet at a level appropriate for detecting and defining the biological effects of human activity (Karr & Chu 1999). The incorporation of more than one class within a dataset facilitates the testing of whether each class holds different vegetation, but requires that enough samples of each class are collected in order to facilitate a search for the influence of anthropogenic impacts within each class (Fore 2003).

A number of different types of wetlands are recognized, differentiated by physico-geographic and hydrological or hydrogeomorphic (HGM) parameters (Brinson 1993). The parameters that differentiate between HGMs, namely landscape and related hydrology, creating different slopes, depths and hydroperiod, are all important determinants of particular types of habitat for plants (Keddy 2000). Geological and climatic constraints are recognized as other important determinants of habitat type and character and the resultant geographical distribution of plants (*sensu* Walter 1973). Collectively, these HGM, climatic and geological parameters are thought to partition a landscape into areas of different habitat for wetland vegetation (*e.g.* Mucina *et al.* 2006a, SANBI 2009). In the context of Southern Africa, it is not yet clear which of these parameters is important in determining the microclimate that exists within the habitat in which wetland plants grow (Cook 2004, Mucina *et al.* 2006a), although alteration of vegetation along a moisture gradient (i.e. hydrological zonation) is better understood (Kotze *et al.* 1994 and 1996, Kotze & O'Connor 2000, Ellery *et al.* 2003). The focus of the current study was thus not restricted to wetlands of a particular HGM type and rather focused on the broader definition of inland freshwater wetlands within planar lowland

landscapes, whilst taking note of the hydrological zonation within which plants were sampled. This is considered as a starting point from which to determine the important criteria for subdivision of different habitat units for wetland vegetation.

One of the challenges of defining the scope of phytoassessment tools is to assess the wetland hydrogeomorphic types or habitats within them for which these tools will be applicable, and under what conditions. Whilst the focus of the current research was largely on inland, freshwater depressional wetlands dominated by emergent plants, there is a gradation of similar habitats between the different wetland types, be they depressions, micro-depressions typical of vernal pools, coastal lakes or riverine backwater depressions. The seasonally waterlogged habitat (or supralittoral zone as described in Section 1.5.2) is a common unit of habitat surrounding most of these wetland types and it is thus hypothesized that vegetation attributes of this habitat may be common to multiple HGM types. Comparison of the seasonally waterlogged (supralittoral) vegetation from multiple HGMs within a group of wetlands for which all other habitat drivers are otherwise similar would facilitate a test of whether this habitat holds similar floristic structure in different HGMs. In the present study the existence of too great a degree of natural variation, as caused by climatic or edaphic differences, between wetlands from different HGMs sampled, may reduce the ability to test this hypothesis.

1.5.1 Comparability of physically different wetland vegetation habitat units

It is important to keep in mind the spatial scale used to differentiate between "habitat units". Whilst wetland represents a different habitat from dryland, within either of these divisions many further subdivisions present different habitat for organisms. Within what are defined as wetlands, saturated soils that can be either oxic or anoxic, inundated surfaces, and the water column each represent a separate growth mediun for vegetation. Each of these media have specific limitations that determine what taxa are able to grow and are considered as relatively distinct habitats (e.g. Keddy 2000, Santamaría 2002, Cook 2004, Smith *et al.* 2007). The combination of flow, drainage, depth and periodicity of the hydroregime are some of the hydrological parameters that determine which organisms can exist, recruit and persist (e.g. Keddy 2000, Kotze & O'Connor 2000). The duration of saturation (waterlogging of the soil pore space) and the duration and depth of inundation (covering of the soils surface by water) are integral determinants of intra-

wetland plant species distribution (*e.g.* Keddy 2000, Mitsch & Gosselink 2000, Cook 2004, Mucina *et al.* 2006a). Hence an array of potential habitats may exist within any wetland, each of which may support plant species with different habitat requirements. Whilst some species, such as many aquatics, will occur only in inundated habitat or the water column, other species have broader environmental tolerances and may occur across these different hydrological habitats (Cook 2004). For phytoassessment purposes, therefore, each habitat provides a potential source of different species that can inform our assessment of environmental condition (US EPA 2002b).

Within each wetland a critical level of habitat separation that is dealt with by the South African National Wetland Classification System (NWCS) (SANBI 2009) is the differentiation of periodicity and depth of inundation and periodicity of saturation brought about by hydrological zonation. The nomenclature used for differentiation between hydrological zones in wetlands within the NWCS (temporary, seasonal, permanent) is not considered useful for differentiation of vegetation habitat as land can be permanently saturated (soil pore spaces are waterlogged) but never inundated whilst another section of habitat may be permanently saturated and occasionally inundated, and another may be permanently inundated. Each of these sections is considered by the NWCS as permanent wetland, yet each constitutes very different habitat for vegetation (Cook 2004). An alternative wetland plant habitat unit nomenclature is presented below.

1.5.2 Hydrological zonation in wetlands creates different habitat for plants

Ground-rooted individual plant specimens do not move; therefore they have to be able to deal with all environmental conditions that exist at a given location throughout the various seasons of the year. Wetlands or the hydrological zones within them can be permanently, seasonally or temporarily wet, thus presenting habitat that ranges, over the course of a year, from permanently inundated (or covered with a water column) through varying degrees of waterlogging of the soil pore space (saturation) to a desiccated terrestrial state (Jones 2002, Ewart-Smith *et al.* 2006, SANBI 2009). The exact delineation of such habitat is not simple because water levels tend to fluctuate (DWAF 2003). Depth of the water column in aquatic environments or in the soil varies with rainfall, infiltration and evapotranspiration rates and hence the spatial hydrological zonation changes with the seasons. From the perspective of wetland vegetation three broad hydrological habitat categories (US EPA 2002b) namely supra-littoral, littoral and

aquatic, are distinguishable as a result of the different hydro-dynamics or residence time of water in and/or inundating the substrate of wetlands. In Figure 1.2the proposed hydrological habitat zonation for seasonal wetlands of the Cape coastal lowlands are presented showing their association with the more commonly used zones of permanent, seasonal and temporary waterlogging (*sensu* SANBI 2009). Aquatic habitat is thus associated with permanent and/or long term inundation, shoreline or littoral habitat is associated with emergence from waterlogged or inundated land, and supra-littoral habitat is depicted In association with seasonally to temporarily saturated and/or inundated land that is prone to long periods of desiccation. All three hydrological habitats may not be present in all wetlands, depending on the type of substrate and the availability of water.

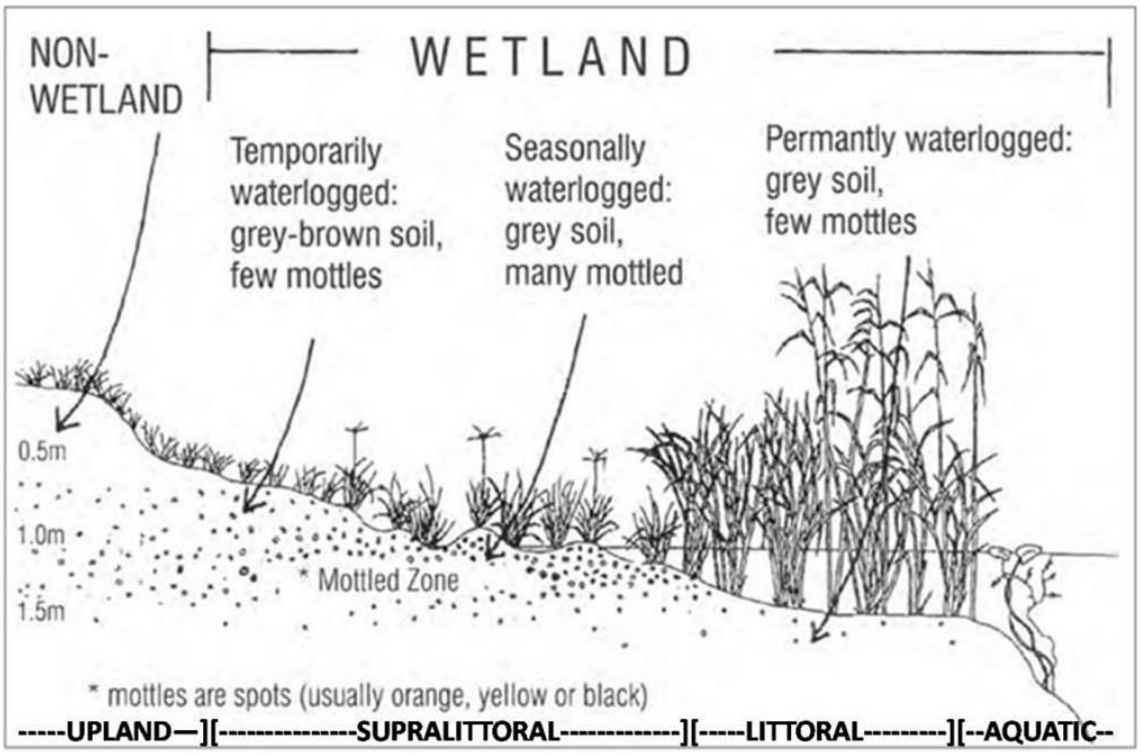


Figure 1.2: Cross section through a wetland, indicating how soil wetness and vegetation indicators change along a gradient of decreasing wetness, from the middle to the edge of the wetland (After Kotze *et al.* 1994 & 1996, DWAF 2003 and with the addition of upland to aquatic habitat separators).

Whilst the littoral is typically associated with marine and lacustrine ecosystems, no freshwater term has been coined to describe the supralittoral habitat. Vegetation within these habitats is described as being either terrestrial, or as submerged within or emergent from the water column, or as aquatic (e.g. Cook 2004), yet these terms are frequently used to describe species that exist in any of the supralittoral, littoral and or aquatic habitats and do not facilitate differentiation between them. Initial field surveys

and discussions with wetland specialists and an examination of the National Wetland Classification System (SANBI 2009) suggested that significant hydrological zonation and concomitant habitat differences were apparent in wetlands of the Western Cape. Although some species may be able to tolerate all zones (Cook 2004), these distinct habitat units are anticipated to support very different communities of plants as a result of hydrological and related environmental differences. The following habitats descriptions were used in the present study to define different hydrological zones:

i. Supralittoral zone

Supralittoral land that is typically ephemerally to seasonally saturated has, if at all, only short periods of shallow inundation and is therefore dominated by vegetation that does not emerge from water standing above the surface of the ground (except for short periods of time) but whose root systems or propagules do at times have to cope with anoxic conditions (caused by waterlogging of the soil pore space). This habitat is typically situated above the line of seasonal to permanent inundation (i.e. above the littoral) and is thus called supra-littoral for the purposes of this study and covers both the temporarily-saturated and part of the seasonally-saturated or waterlogged sections depicted in Figure 1.2. Supralittoral land supports hydrophytes or plants that are adapted to waterlogging and live either in water itself or in very moist soils. Such plants include perennial terrestrial taxa typical of marshes or vleis that are tolerant of submergence (helophytes), and many annual plants that survive the unfavourable season (dry season) in the form of seeds and complete their life-cycle during favourable seasons for growth (therophytes) (*e.g.* Raunkiær 1904,Cook 1996). The hydrology of this habitat is considered akin to that of the wet meadow habitat described by Galatowitsch *et al.* (2000) and DeKeyser *et al.* (2003); but as the supralittoral habitat is not fed by snow melt or long-distance drainage it is considered closer to that of the Vernal Pool habitat of mediterranean ecosystems described by Keeley & Zedler (1996), in which soils are prone to summer desiccation.

ii.) Littoral zone

The littoral zone is land that is permanently waterlogged and has seasonal to permanent inundation, that is dominated by so called emergent vegetation typically associated with the shore line as an interface between an aquatic and a terrestrial habitat. Littoral land supports helophytes, tenagophytes (plants with the juvenile phase submerged in or floating on water and the adult [flowering] phase terrestrial) and hyperhydates (plants with

leaves and or stems emerging above the water surface) (Cook 2004). All of these plant forms have submerged, and/or emergent and in many cases terrestrial stages in their life histories.

iii.) Aquatic zone

The aquatic zone is land that is permanently to seasonally inundated to a depth (typically greater than two metres) that excludes substrate rooted species whose reproductive parts need to emerge from the water column (e.g. *Typha capensis, Phragmites australis*). The aquatic zone provides a water column and inundated substrate for an extended period of the growing season, supporting aquatic vegetation, which is otherwise dormant during terrestrial periods of draw-down.

1.5.3 Summary of comparable vegetation units

In terms of the phytoassessment of environmental condition, habitat specificity of plants allows comparison between similar habitat units from different wetlands. The total complement of a wetland plant community is a reflection of all the different available habitats within a wetland. This suggests that comparison of the entire community of plants between two or more wetlands only makes ecological sense when the wetlands being compared contain the same habitats. For phytoassessment development purposes, testing whether the supralittoral vegetation of wetlands is distinctly different from that of the littoral habitat would thus allow us to know whether to compare the entire vegetation complement of wetlands or only to compare that of individual hydrological zones. In essence, even in the absence of anthropogenic stressors, the occurrence of different habitats within a single wetland means that a mosaic of different plant communities would be present within each wetland (Santamaría 2002, Cook 2004). Hydrological zones and hydroregimes, texture of the substrate, nutrient loads, pH levels, phytogeographical regions, and plant structural types (architecture) are all discriminators that have been used for separation of comparable units of wetland habitat and vegetation (Denny 1985, EUR15 1999, US EPA 2002b, Cook 2004, Mucina *et al.* 2006a, Ewart-Smith *et al.* 2006, SANBI 2009).

In summary, the separation of vegetation into units from different nutrient and substrate classes and hydrological habitats, and different 'structural' (plant-architecture such as herbaceous, shrubby or forested) units in different climatic and geological regions can

provide units of plant species that can be expected to respond similarly to a given anthropogenic or natural stressor.

1.5.4 Plants as obligate and facultative wetland taxa

The distinction between a dryland plant and a wetland plant, which is a standard indicator used in the DWAF (2003) wetland delineation protocol, is also used in many other countries for delineating wetland boundaries, in wetland inventories and for regulatory purposes (Tiner 1991, 1999). For delineation purposes wetland plants have been split into two groups (Reed 1988):

- Obligate wetland plants - dependent on wet or anaerobic conditions for growth, and
- Facultative wetland plants - capable of growing in anaerobic soils but also reasonably competitive in well-aerated soils.

Observation of plant distributions within wetlands does not facilitate the discrimination of whether a plant is a hydrophyte and thus restricted to wetland habitat and obligated to occur within such environment or if it is a facultative wetland plant that is essentially terrestrial or a helophyte, and is able to survive waterlogging and occasional inundation (Cook 2004). Such discrimination would be best provided by physiological examination of plant structure and extensive and accurate records of distribution and ecological conditions of the habitat in which species are recorded. This information is severely lacking in the South African context (Cook 2004, Mucina *et al.* 2006a). A pragmatic approach to the designation of whether a plant has facultative or obligate association with wetland as based on frequency of occurrence (correlation) in wetland relative to dryland habitat was adopted by the US Fish and Wildlife Service (Reed 1988) (Table 1.2).

Table 1.2: Classification of plants according to occurrence in wetlands as based on U.S. Fish and Wildlife Service Indicator Categories (Reed 1988).

Category	Abbrev'n	Category description
Obligate wetland species	O	Almost always grow in wetlands (>99% of occurrences)
Facultative wetland species	Fw	Usually grow in wetlands (67-99% of occurrences) but are occasionally found in non-wetland areas
Facultative species	F	Are equally likely to grow in wetland and non-wetland areas (34-66% of occurrences)
Facultative dryland species	Fd	Usually grow in non-wetland areas but sometimes grow in wetlands (1-34% of occurrences)
Dryland species	D	Almost always grow in drylands (>99% of occurrences)

Given that the discrimination relies on correlation, for any given plant, the accuracy of this designation therefore relies on the number of observations made of that taxon. In South Africa, a list of wetland associated taxa was compiled based on habitat descriptions on labels and vouchers of herbarium specimens (Glen *et al.* 1999, Glen unpublished). This national list of wetland associated taxa includes a designation as obligate or facultative but does not include the number of observations that this status was based on. A plus or minus sign adjacent to any of the categories was used by Glen to indicate greater or lesser conviction of the status for any given species. Glen's list of taxa along with the addition of further wetland associated species in the Cape Flora as extracted from the habitat descriptions provided by Goldblatt and Manning (2000) are appended to this thesis in Appendix 6, a compact disc of sample data.

The concept of facultative and obligate taxa has not proven useful in determining environmental condition in the ambit of bioassessment (U.S. EPA 2002a). The alteration of hydrological regimes have well documented impacts on species groups: plant zonation patterns shift as plants intolerant of the hydrologic alteration and the new hydroregime are replaced by tolerant species and terrestrial species may invade or die back due to drainage or flooding (Adamus *et al.* 2001). It is clear, therefore, that a decrease in the number of obligate wetland taxa and an increase in facultative species or non-wetland species can result from hydrological changes such as drainage or excessive water abstraction. Determining whether hydrological zones have shifted, beyond temporary levels of change, when there are no obvious signs of dehydration (such as soil mottles in a zone that is no longer saturated) or of increased hydration (such as the absence of upland taxa) is extremely difficult to determine in seasonal wetlands from a single or even multiple field visits over a limited time span (Brooks 2004) and was not attempted in the present study.

1.5.5 Life-history groups

In bioassessment using invertebrate taxa, different genera, families or functional feeding groups are often used to separate taxa with different resource needs and thus differential affiliation with different environmental conditions (e.g. Helgen & Gernes 2001, Bird 2010). The hierarchy of plant taxonomic groups does not lend itself to a similar use; a good example is the genus *Senecio*, which contains plants that range from small, fleshy herbs

to large woody shrubs adapted to a range of habitats from wet to arid. *Senecios* are a member of the family Asteraceae, the taxa of which are also extremely morphologically diverse and occur in a vast array of habitats. Similarly, while invertebrates are separated into feeding guilds (Root 1967), plants typically all compete for similar resources (Harper 1977, Grubb 1977). Hence different methods of separating them have been developed, based on the different means by which plants acquire resources (e.g. Grime 1979). Some functional groups of plants have been based on physiological and morphological traits associated with functional strategies employed in the use or acquisition of resources (e.g. Hutchinson 1975, Grime 1979, Boutin & Keddy 1993, Cook 1996). Similarly but with greater emphasis on life-history traits [annual/perennial], origin [alien/indigenous], and broad growth form [graminoid/herbaceous/woody] taxa have been grouped for phytoassessment purposes in wetlands by the work of McIntyre *et al.* (1995), Galatowitsch & Van der Valk (1996) and Galatowitsch *et al*. (2000). These latter so-called life-history groups provide intrinsically simple means of grouping species with similar traits.

Whilst species and associated community assemblages may differ between wetlands of different classes and substrates, life history groups (*sensu* Galatowitsch *et al.* 2000) may provide a means of comparing the impacts of land-use within multiple classes of wetlands. The use of life history groups (or guilds) to represent the species in wetlands may therefore assist in the endeavour to develop methods for phytoassessment over multiple habitat types or even multiple phytogeographical regions (Galatowitsch *et al.* 2000). Furthermore these groups of taxa provide potential units within which to search for like response or characteristic affinity for reference and impaired conditions.

1.6 Objectives of the study

The intent of the present study was to develop phytoassessment tools for the evaluation of environmental condition of wetlands in the coastal lowlands of the south-western Cape. In the anticipation that wetlands with Cape Lowland Freshwater vegetation (Mucina *et al.* 2006a), within the supposedly homogenous phytogeographical Western Coastal Slope region (Cowan 1995), should be considered a unit of comparable wetland vegetation habitat fifty nine such wetlands were sampled. The phytosociology of this wetland vegetation and associated environmental data were examined in order to identify geographical units of land that do in fact contain comparable wetland habitat and thus a relatively homogenous community of wetland vegetation. Within such units of land an

empirical comparison of impaired *vs.* minimally impaired (reference) wetland plant communities was performed in order to identify attributes of the vegetation assemblage that are consistently and characteristically associated with different environmental conditions. Species and vegetation attributes with characteristic association to environmental conditions, can each potentially be developed into a measure or metric of environmental condition. Metrics can be amalgamated into a multi-metric phytoassessment index, thereby facilitating a quantitative determination of environmental condition.

The unusual heterogeneity of the environment of the Fynbos Biome provides an opportunity to study the composition of wetland vegetation from numerous habitats in close proximity to one another. Findings may inform our understanding of the macrocosm of considerable diversity of wetland habitats available in South Africa. This study also provides an opportunity to evaluate:

- the phytogeography and suggested azonality of wetland vegetation (that hydroregime exerts an influence greater than the macroclimate on floristic composition (Mucina *et al.* 2006a)); and
- the environmental drivers of difference between habitat units and the applicability of the discriminators used by the National Wetland Classification System (SANBI 2009) for separation of different units of wetland habitat (as summarised in Section 1.5.3).

For the Cape coastal lowlands of South Africa this study represents a starting point to determine:

- The geographical area (*sensu* wetland regions or subsets thereof) within which wetlands have similar species assemblages i.e. whether there is considerable and significant beta diversity difference between different sub-regions, bioregions or associated terrestrial units of vegetation within the chosen region;
- Whether wetland plants, within this region, or sub-regions thereof, have affiliation with different degrees of environmental condition as caused by human disturbance; and thus
- The identification of potential indicator species or groups of species and other comparative measures of community assemblage such as evenness and diversity, which are representative of different categories of human disturbance.

The field work for this study was carried out before the SANBI (2009) version of the National Wetland Classification System was completed and helped to guide the decision to incorporate the discriminator that would differentiate between units of vegetation within the NWCS. Hence, whilst field work for the thesis was carried out with only the guiding discriminators of the Ewart-Smith *et al.* (2006) study the new classification provided by SANBI (2009) was used to group like habitats for the thesis and these discriminators are thus considered to be tested by this research. This study is specifically focused on the ecology of inland, freshwater wetlands in planar landscapes with lentic conditions, endorheic drainage and seasonal to permanent hydroregimes in the Cape coastal lowland. As the wetlands for this study were sampled with the intent of determining phytoassessment tools, both impaired and minimally impaired wetlands were sampled.

1.6.1 Spatial Scales of investigation

The objectives of the study can be anchored at three different spatial scales:

a) At a Fynbos Biome level, I investigated the phytogeography and azonality of freshwater wetland vegetation in western, central and eastern sub-regions of the mediterranean Western Coastal Slope wetland region (SW.m) of Cowan (1995) on the Cape coastal lowlands. Investigation of the influence of macroclimatic and geological parameters on phytogeography was performed by a comparison of the community structure of wetlands from different bioregions created by Rebelo *et al*. (2006). A comparison of the distribution of species relative to spatially related environmental variables was also performed.

b) At a regional level (within western, central or eastern sections of the Cape coastal lowlands) I investigated the distribution of freshwater wetland vegetation as delineated by association with the geological substrates of the zonal terrestrial vegetation units, thus testing the hypothesis that wetland vegetation can be further classified into different vegetation units based on underlying geology (e.g. sand, shale, ferricrete *et cetera*) as proposed by Cowan (1995). Verifying or refuting this hypothesis would enable conservation planners to make informed decisions on classification, management and phytoassessment development for freshwater wetland vegetation units of Cape coastal lowland areas.

c) At a landscape level I investigated:

- the role of hydrological zonation (supralittoral/littoral/aquatic) on intra-wetland floristic community dynamics. This distinction may also be applicable at all previous spatial scales; and
- the role of anthropogenic disturbances on intra-wetland floristic community structure within a phytogeographically homogenous unit of wetland vegetation.

1.7 Structure of this thesis

This thesis is divided into six chapters namely introductory and methods chapters, followed by three chapter's dealing with different aspects or spatial scales of the collected data and a concluding chapter.

Chapter 1 provides a literature review of the assessment of environmental condition of seasonal wetlands using vegetation and expounding the philosophical approach used in researching the potential for development of phytoassessment tools for wetlands of the coastal lowlands of the south-western Cape of South Africa.

Chapter 2 deals with the study sites and assessment methods adopted in the present study.

Chapter 3 is an examination of the diversity and geographical distribution of wetland vegetation in the mediterranean Western Coastal Slopes wetland region of the Cape coastal lowlands.

Chapter 4 attempts to identify the correlation between the anthropogenic, climatic and geological drivers of species distribution patterns within any unit of wetland vegetation with a relatively homogenous set of taxa.

Chapter 5 attempts to identify species, life history groups, or other vegetation attributes with characteristic association for impaired relative to minimally impaired wetlands of a given unit of vegetation.

Chapter 6 provides a general discussion of the outcomes of this research and their implications for management and conservation of wetlands in the Cape coastal lowlands and the rest of South Africa. The practicality of developing phytoassessment tools is discussed in the light of these findings and some guidelines are provided that would assist with further attempts to develop phytoassessment indexes for South African wetlands.

CHAPTER 2

STUDY SITES & METHODS

A brief description is provided of the study sites as embedded within the Western Coastal Slope region of the Cape coastal lowlands of the Fynbos biome. The methods described are those implemented to facilitate the collection of vegetation sample data, human landuse, and ambient environmental conditions, in order to test hypotheses regarding the relationship between macrophytes, their spatial distribution and the environmental condition of their habitats.

2.1 Study sites

This study was focused south of 32^0S and west of 20^0E and below 200m a.m.s.l. in the south-western corner of the Western Cape Province of South Africa (see Figure 2.1). This area of focus was within the Fynbos Biome (*sensu* Rebelo *et al.* 2006), which is a subset of the area of the Cape Floristic Kingdom The wetlands assessed in this study are situated on the lowland plains of the coastal forelands, the physiographic zone between the Cape Fold Mountains and the coast (Lambrechts 1979). In this study the coastal forelands are called the Cape coastal lowlands. The topography of these coastal lowland plains is predominantly flat with low to moderate relief (e.g. Lambrechts 1979, Kleynhans *et al.* 2005). For the wetlands that were assessed, altitude ranged from 5 to 120 metres above mean sea level (Google Earth, accessed 2007). The wetlands assessed are from within the Western Coastal Slope region (Cowan 1995), as discussed in Section 1.4.4.1, which is a subset of the total area of the Cape coastal lowlands.

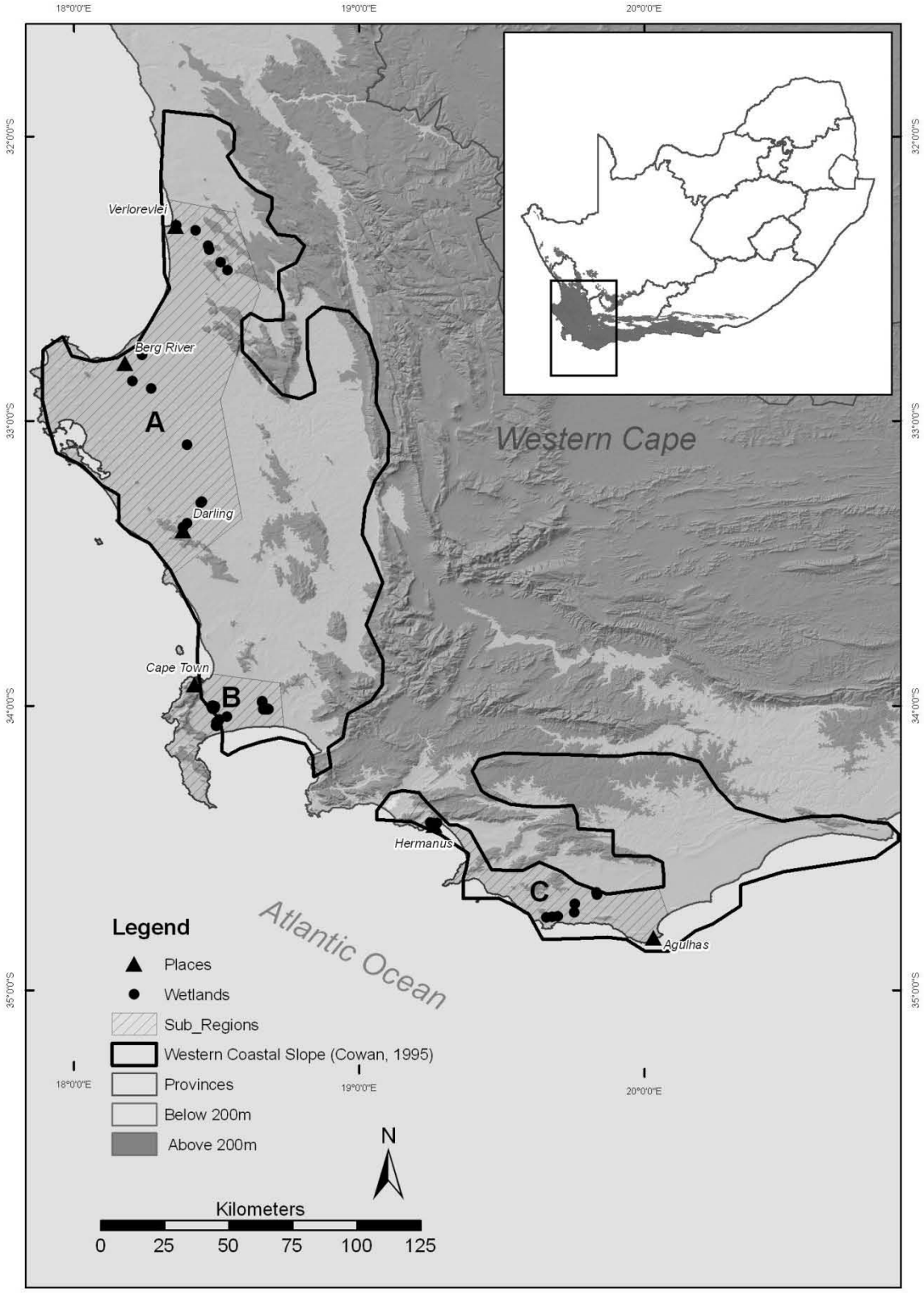


Figure 2.1: Inset is a map of South Africa and its provincial boundaries showing the shape of the Fynbos Biome. The lighter shaded area in the large map depicts the land below 200 meters above mean sea level in the south-western corner of the Fynbos Biome. The outline of the two sections of the mediterranean Western Coastal-Slope wetland region of Cowan (1995) are marked. The position of wetlands and clusters of wetlands sampled in different areas of this region are marked on the map. The western (West Coast), central (Cape Flats) and eastern (Overberg) sub-regions are marked on the map as indicated by A, B and C respectively.

The geographical unit of the Cape coastal lowlands below 200m a.m.s.l. depicted in Figure 2.1 incorporates the mediterranean climatic zone of the South Western Coastal Slope region (SW.m) (Cowan 1995) (See Figure 1.1 in Chapter 1 to see all of Cowans wetland regions). In Figure 2.1 a digital transformation of Cowans two SW.m units were overlaid onto the map of the Western Cape using ArcView in GIS. The discrepancy between the boundaries of the SW.m areas and that of the coast area are a result of the rough transformation of the regions of Cowan and the resolution at which these original regions were outlined. The full extent of the SW.m region is contained within a smaller area than the geographic extent of the Cape Lowland Freshwater vegetation unit as mapped by Mucina *et al.* (2006a). Any vegetated freshwater wetlands that were within the SW.m and that fulfilled the classification criteria chosen from the National Wetland Classification System (SANBI 2009) were considered as possible sampling sites (See Section 2.2.3 below).

Subsets of wetland plant communities were sampled in the three spatially disparate sub-regions of the SW.m (Cowan 1995), namely on the West Coast, the Cape Flats and the Overberg as shown in Figure 2.1 and individually depicted in Figures 2.2, 23 and 2.4. In each of these sub-regions the wetlands sampled are shown in the context of the surrounding terrestrial vegetation unit (Rebelo *et al.* 2006) and, in situations where they were mapped by Mucina *et al.* (2006a), also in the context of the wetland vegetation unit. Wetlands occur in close proximity in these three sub-regions within which the intensity and extent of human landuse varies considerably (De Roeck 2007). Within each sub-region there was further concentration of sampling effort in distinct localities where wetlands were most abundant due to topography and relief. Assessment effort was concentrated in three sub-regions with the dual intentions of:

- minimizing potential changes brought about by environmental differences within each sub-region; and
- assessing the species turnover (change in beta diversity) across the Western Coastal Slope region, as well as across each sub-region, and even within sub-regions where necessary.

The question needs to be asked whether, within comparable habitats, considerable difference (turnover) in plant diversity is apparent from one sub-region to the next? Considerable difference between the sub-regions in the percentage of wetland plant species that are unique to one sub-region would indicate the existence of considerable turnover or beta diversity differences. Fifty nine wetlands were sampled with the intention of being able to develop metrics for the Western Coastal Slope region, in the anticipation that wetlands with Cape Lowland Freshwater vegetation in this region should be considered a single unit of comparable wetland vegetation.

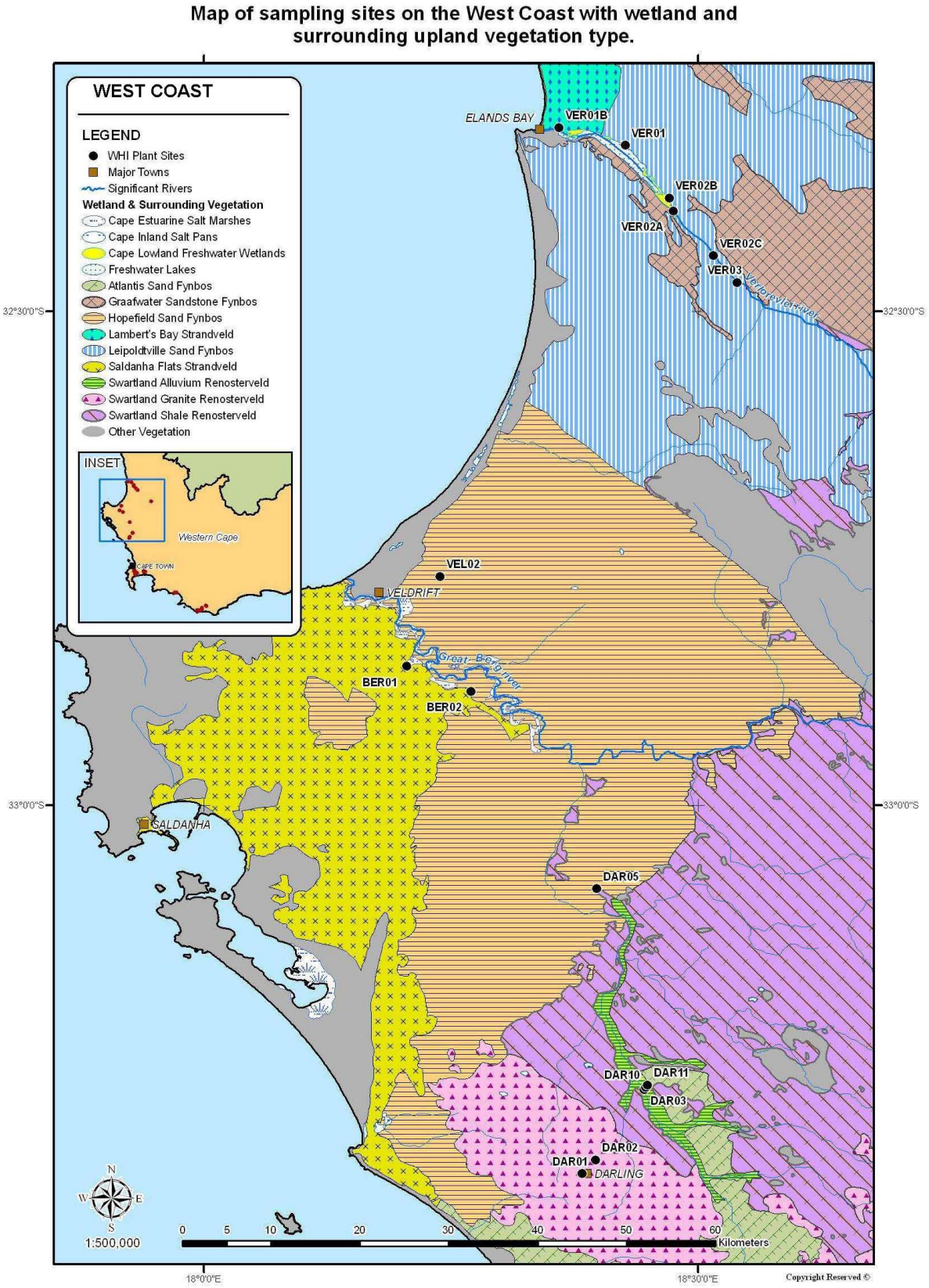


Figure 2.2: The western sub-region of the Cape coastal lowlands: A or West Coast from Figure 2.1. The wetlands sampled are indicated within the associated terrestrial and/or wetland vegetation units as mapped by Rebelo *et al.* (2006) and Mucina *et al*. (2006a) respectively. (The towns of Veldrift and Elandsbaai were respectively marked as Berg River and Verlorevlei in Figure 2.1)

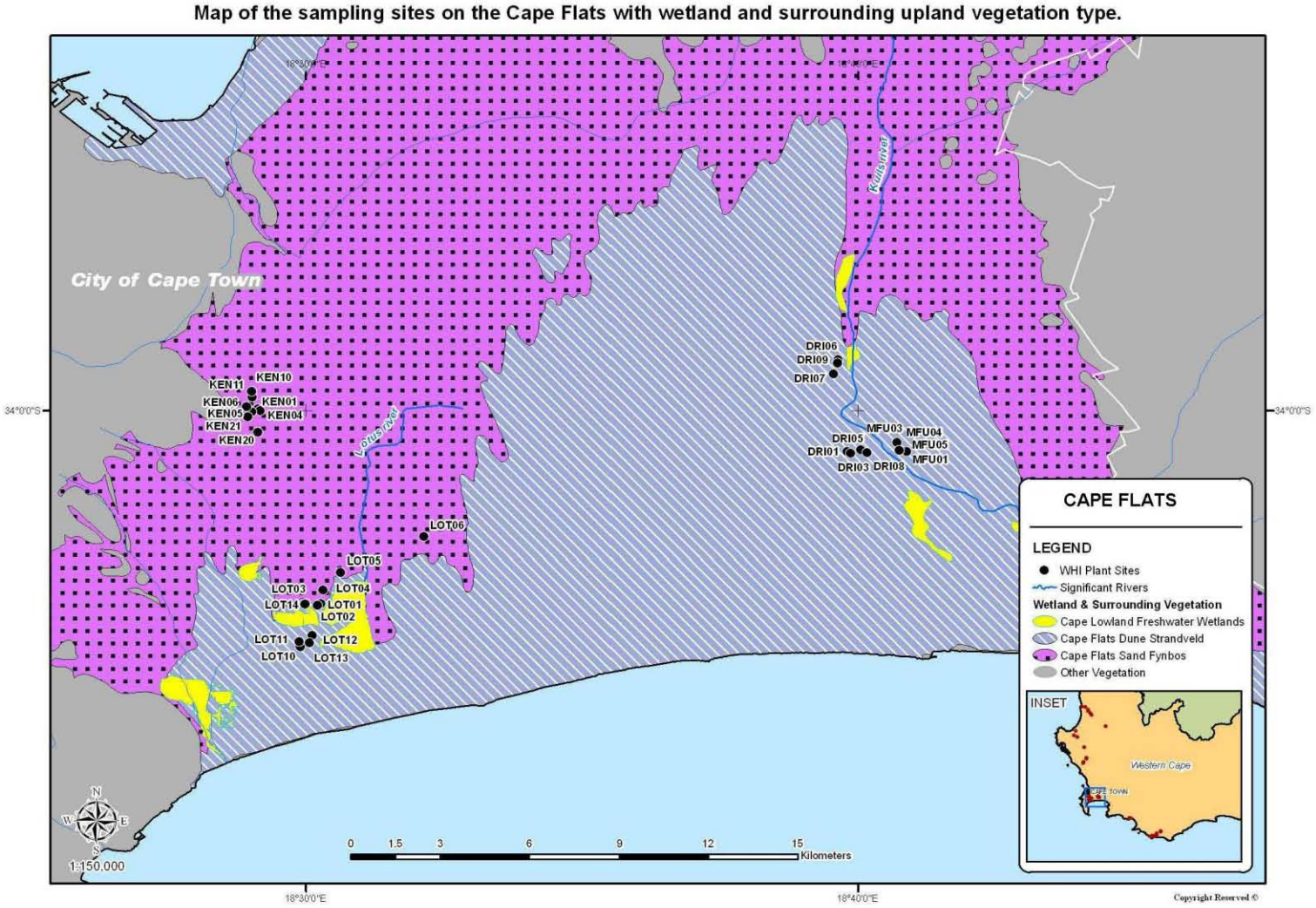


Figure 2.3: The central or Cape Flats sub-region of the Cape coastal lowlands (section B from Figure 2.1). The wetlands sampled are indicated within the associated terrestrial and/or wetland vegetation units as mapped by Rebelo *et al*. (2006) and Mucina *et al*. (2006a) respectively.

Map of sampling sites on the Overberg coastal plain within the surrounding wetland and upland vegetation types.

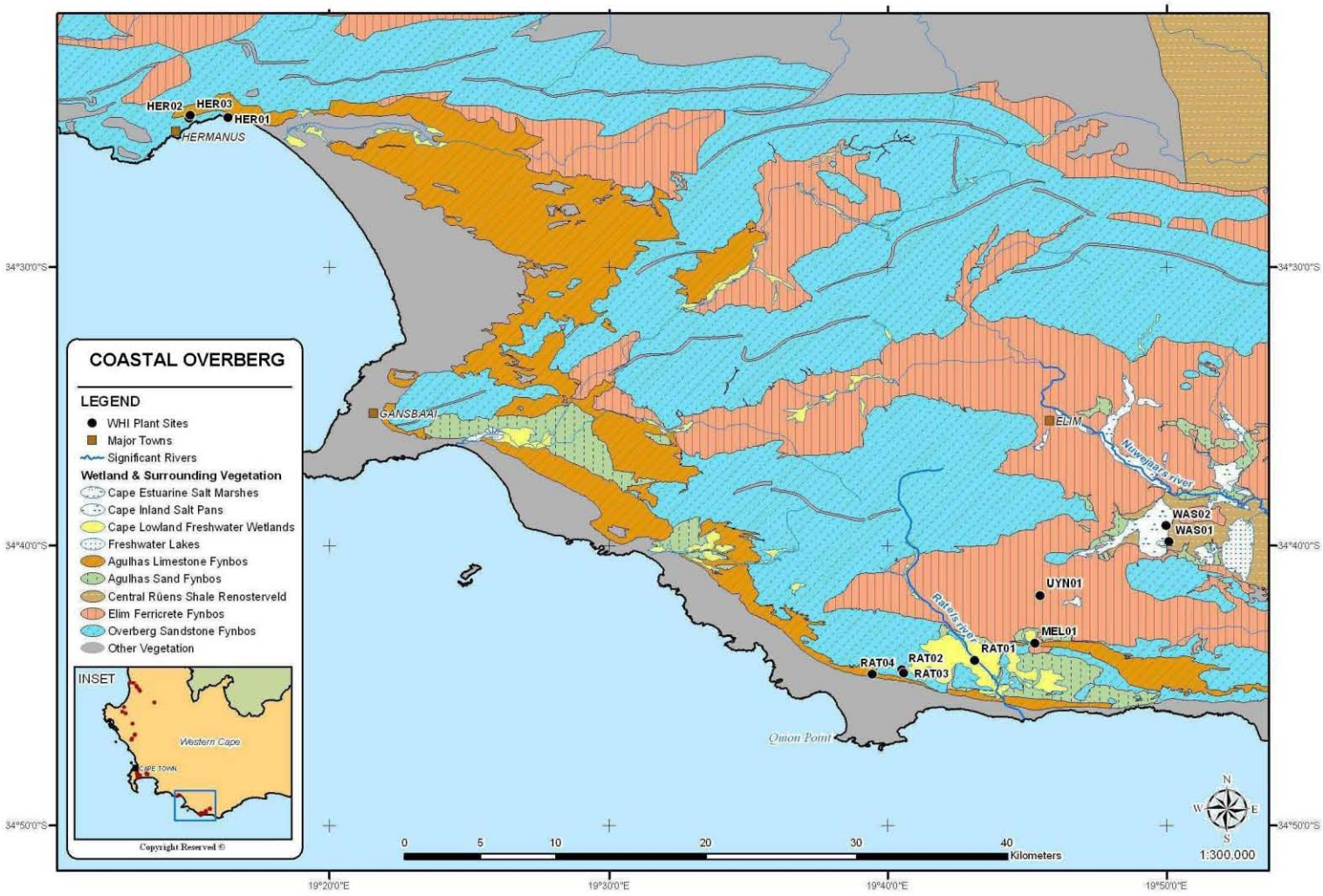


Figure 2.4: The eastern sub-region of the Cape coastal lowlands: C or Overberg from Figure 2.1. The wetlands sampled are indicated within the associated terrestrial and/or wetland vegetation units as mapped by Rebelo *et al.* (2006) and Mucina *et al*. (2006a) respectively.

Cainozoic (Tertiary and Quaternary) deposits of calcareous sands form the geological basis of this area, within which limestone deposits are an occasional feature (Cowan 1995). Levels of soil nutrients, in particular nitrogen and phosphorus, are generally very low, and are considered to act as determinants of the community assemblage of the terrestrial or zonal (as opposed to wetland) vegetation (Specht & Moll 1983, Kruger *et al.* 1983, Cowling *et al.* 1997, Rebelo *et al.* 2006). Geological and climatic differences within this region drive the expression of soil nutrient and salt concentrations, resulting in different sets of terrestrial vegetation types across the region (Rebelo *et al.* 2006, See Appendix 1).

The study area is characterised by a mediterranean-type climate, not found elsewhere in sub-Saharan Africa, with cool wet winters and relatively dry and warm summers that have been in place since the Late Pliocene (*circa* 3.2-2.5 M.yr) (Deacon *et al.* 1992). At the low altitude that this study was conducted at (below 200m a.s.l.) the mediterranean climate results in mild winters with no frost and minimum temperatures that do not terminate plant growth. Aridity increases from the Agulhas Plain in the south-east to the Cape Flats and West Coast in the north-west (from C to A in Figure 2.1) (e.g. Rebelo *et al.* 2006). An increasingly seasonal concentration of rainfall on the West Coast relative to the Agulhas Plain, calculated from the duration and extent of mean winter rainfall as a percentage of mean annual rainfall, exacerbates this trend (Rebelo *et al.* 2006). A summary of some of this information is provided in Table 2.1.

Table 2.1: Environmental characteristics of the three sub-regions in which wetlands were investigated

Area	West Coast (A)	Cape Flats (B)	Overberg: Agulhas Plain + Hermanus (C)
No. Wetlands investigated	16	33	7 + 3
Altitude range[1](m)	2-120	5-40	10-65
Precipitation[2] annual average & [5] (average all sites) ($mm.a^{-1}$)	200-500 (317)	500-800 (761)	500-800 (565)
Evaporation[2 & (5)] (annual average) & (average all sites) ($mm.a^{-1}$)	2000-2500 (2300)	2000-2500 (2000)	<2000 (1800)
Moisture balance[2] ($mm\ month^{-1}$)	Summer (Jan'y) -350 Winter (July) -25	Summer (Jan) -200 Winter (July) +50	Summer (Jan'y) -240 Winter (July) 0
Geology[3]	Cainozoic sandy and calcareous coastal deposit	Cainozoic sandy and calcareous coastal deposit	Cainozoic sandy deposits + Mio-Pliocene shallow water limestone & coastal sandy deposits
Surrounding zonal vegetation bioregions[3]	South West Fynbos, West Strandveld & West Coast Renosterveld.	South West Fynbos, West Strandveld	South West Fynbos, South Coast Fynbos & East Coast Renosterveld
Land-use[4]	Cultivated, grazed and fallow land and 'industrial urban edge'	Residential + 'industrial urban', agricultural and Conservation Area	Abandoned agricultural land and Conservation Area + Golf estate on urban edge

Citations: [1] Google Earth (2007); [2] Deacon *et al.* (1992); [3] Mucina *et al.* (2006a); [4]: Field assessment, [5] Schulze (2006).

i. West Coast climate, geology and vegetation

In the area broadly stretching from Verlorevlei southward to Darling, the West Coast has a semi- to sub-arid climate. The area has a strongly (80%) winter-concentrated wet season, with less than 250 mm mean annual precipitation around Verlorevlei (Schulze 2006). The geology is dominated by marine and thus calcareous sands of high base (Mg^{2+}, Ca^{2+}) status. Soils of the West Coast have mostly developed from recent drift sands. Near the coast the sands are highly calcareous whilst inland the lime content gradually decreases through leaching (Rebelo *et al.* 2006). The terrestrial vegetation of the West Coast has large swathes of each of the three broad vegetation constituents of

the Fynbos Biome, namely Fynbos, Strandveld and Renosterveld (Rebelo *et al.* 2006). Wetlands with 'Cape Lowland Freshwater' vegetation, 'Vernal Pool' vegetation and inland salt pans with their associated 'Cape Inland Salt Pan' vegetation unit all occur in this area (Mucina *et al.* 2006a) (See Figure 2.2). On the West Coast wetlands were sampled at Darling, and along the Berg River, and Verlorevlei at Elandsbaai. In these localities, predominantly impacted by agricultural land-uses, nine isolated depressions and a further seven non-isolated wetlands were sampled, six of which were separate areas within the Verlorevlei wetland.

ii. Cape Flats climate, geology and vegetation

The Cape Flats is less arid than the West Coast, but is also characterized by winter rains (Schulze 2006). The geology is dominated by sandy coastal deposits, with low base status in acid soils and higher base status in alkaline soils (Deacon *et al.* 1992). Cape Flats Sand Fynbos and Cape Flats Dune Strandveld dominate the terrestrial vegetation types, respectively reflecting the spatial distribution of acidic and alkaline sandy soils (Rebelo *et al.* 2006). Cape-Lowland-Freshwater-vegetation-dominated wetlands are mapped as distinct units of vegetation in this locality whilst Cape Vernal Pools, which are typically too small to be mapped, also occur in the area (Mucina *et al.* 2006a) (See Figure 2.3). Wetlands from both the acidic and alkaline sands were assessed. On the Cape Flats, in the Kenilworth, Kuils River Floodplain (Mfuleni and Driftsands) and Lotus River areas, a total of 32 depressional wetlands and one wetland flat were sampled. These wetlands ranged from least impaired reference sites to severely urban-impacted ecosystems.

iii. Overberg climate, geology and vegetation

The Overberg is less arid than the West Coast but more so than the Cape Flats and receives predominantly winter rainfall (Schulze 2006). Along the coastal belt the geology of the Overberg consists of sandy deposits with shallow-water limestone and ferricrete (Deacon *et al.* 1992). The terrestrial vegetation of the lowlands of the Overberg is dominated by vegetation of the broad Fynbos, Strandveld and Renosterveld type (Rebelo *et al.* 2006). No wetlands within the Strandveld vegetation type were assessed in the Overberg in the present study. Wetland vegetation units of Cape Inland Salt Pan and Cape Lowland Freshwater occur at a large enough scale to be mapped (Mucina *et al.* 2006a) (See Figure 2.4). In the Overberg, five depressions and two wetland flats were sampled on the Agulhas Plain (least impaired and agricultural-impacted wetlands); and within the urban edge of Hermanus, three seeps with moderate impairment were sampled.

iv. Comparative summary of sub-regions

In summary there are some broad climatic and geological similarities between the three sub-regions. The arid to semi-arid mediterranean climate and predominantly sandy geology result in conditions that commonly support ephemerally- to seasonally-inundated isolated depressional wetlands (Jones 2002). These wetlands are more similar to the Vernal Pools described by Keeley & Zedler (1996) rather than the permanently waterlogged or inundated depressions common to less arid regions of the world. These similarities suggest the potential for each sub-region to contain similar wetland vegetation. The seasonality of the hydrological regime creates distinct hydrological zonation within wetlands of the Western Coastal Slopes, presenting different habitat for wetland plants within supralittoral, littoral and aquatic zones (see Section 1.5.2 in Chapter 1 and 2.2.3 below). A tabular summary of the localities, wetlands and HGM vegetation type combinations sampled in each sub-region is presented below in Table 2.2.

Table 2.2: Spatial hierarchy of wetlands with different vegetation + hydrogeomorphic habitat in localities from each sub-region of the Western Coastal Slope wetland region.

Sub-region	Locality	Types of Impacts	"Wetland vegetation-unit" + "HGM-type"			Terrestrial Vegetation Units (Mucina *et al.* (2006a)
			depressions		Various vegetation + HGM combinations	
			CLF-	Vernal-		
West Coast	Berg River	agricultural	1	1	1 Saline-floodplain	Saldanha Flats Strandveld, Hopefield Sand Fynbos
	Darling	agricultural & urban	3	2	2 Alluvial-floodplain	Swartland Granite Renosterveld, Swartland Alluvium Renosterveld, Hopefield Sand Fynbos, Atlantis Sand Fynbos
	Verlorevlei	agricultural	-	-	3 CLF-Floodplain, 3 CLF-Valley-bottom	Leipoldtville Sand Fynbos, Lamberts Bay Strandveld
Cape Flats	Driftsands	urban	9	2	-	Cape Flats Dune Strandveld
	Kenilworth	urban	9	1	1 CLF-Flat	Cape Flats Sand Fynbos
	Lotus River	urban	11	-	-	Cape Flats Dune Strandveld, Cape Flats Sand Fynbos
Over-berg	Hermanus	recreational	-	-	3 CLF-seeps	Overberg Sandstone Fynbos
	Agulhas	agricultural	4	1	1 Saline-floodplain, 1 CLF-flat,	Overberg Sandstone Fynbos, Elim Ferricrete Fynbos, Agulhas Limestone Fynbos, Central Ruens Shale Renosterveld

CLF = Cape Lowland Freshwater vegetation

A table of the abiotic characteristics of each wetland and the terrestrial vegetation unit surrounding each wetland are presented in Appendix 2.

The decision to sample over such a wide geographical area was based on the consideration that the Cape Lowland Freshwater vegetation unit is considered azonal (Mucina *et al.*2006). The sample size of the present study might not prove sufficient to facilitate the development of robust metrics if too great a variety of habitats and wetland vegetation types were inadvertently included. This all-inclusive approach could however, in theory, yield phytoassessment metrics that may be applicable over the full spatial and wetland habitat range of the data set.

2.2 Methods

The following is a description of the methods used in the present study.

A targeted sampling approach was followed whereby suitable sites were chosen based on accessibility and on rapid field-based assessment of the amount of anthropogenic disturbance as an *a priori* surrogate for overall environmental condition.

2.2.1 Timing of sampling

Sampling was conducted during the spring and early summer of 2007. This period coincides with the flowering and seed formation of graminoid species, and overlaps with the end of the season for the early spring flowering of some geophytic taxa (Goldblatt & Manning 2000).

2.2.2 Delineation of wetland and hydrological zones

Where the wetland-terrestrial boundary was not apparent for any wetland, a rough delineation procedure was carried out by soil augering and by examination of vegetation and indicators of past standing water using the method of DWAF (2003). The delineation process ensured:

- that all samples and observations recorded from each site were made within the wetland rather than in terrestrial areas; that
- the full extent of the wetland area was assessed, including all of the ephemerally to seasonally saturated habitat of the supralittoral zone; and that
- the approximate extents of the supralittoral, littoral and aquatic hydrological zones were determined.

2.2.3 *A priori* habitat classification

To make apparent the differences in plant community assemblage brought about by both human-related stressors and natural environmental differences *a priori* classification of

wetland habitat and assessment of human impact were performed. The following is a description of the classification of distinct units of wetland habitat and this is then followed, in Section 2.2.4, by a description of the assessment of human impairment.

The National Wetland Classification System (initially using discriminators from Ewart-Smith *et al.* (2006) and later replaced by the discriminators in SANBI (2009)) using the wetland vegetation types of Mucina *et al.* (2006a) as incorporated within the wetland regions of Cowan (1995) were used as a starting point to determine comparable habitat units of wetland vegetation. Whilst SANBI (2009) separates regions with potentially similar biota based upon the ecoregions of Kleynhans *et al.* (2005) the wetland regions of Cowan (1995) were proposed for this purpose by Mucina *et al.* (2006a) and are therefore used in the present study. For purposes of clarity the overlap between the ecoregions and wetland regions is briefly addressed. The western section of the mediterranean Western Coastal Slope wetland region (SW.m) (Cowan 1995) overlaps with the South Western Coastal Belt ecoregion of Kleynhans *et al.* (2005), other than for the area around Verlorevlei which Kleynhans *et al.* (2005) place within the Western Coastal Belt ecoregion. The eastern section of the SW.m which is adjacent to the western edge of the temperate region of the Southern Coastal Slope (SS.a) (see Figure 1.1) would, however, be incorporated along with the SS.a in the Southern Coastal Belt ecoregion (Kleynhans *et al.* 2005).

This study focused on comparable habitat units within:

- inland, vegetated, freshwater wetlands with a hydroregime of lentic conditions characterized by a range of ephemeral to permanent saturation and predominantly seasonal inundation, as well as by endorheic or exhoreic drainage (*sensu* SANBI 2009); as situated on
- the Coastal Foreland or Cape coastal lowlands of the Western Cape (Lambrechts 1979), that have a similar position in the landscape (from a planar landform) and a similar hydrogeomorphic setting (*sensu* SANBI 2009) (See Figure 2.1); and encompassing
- Cape Lowland Freshwater vegetation, although a number of Vernal Pools, saline and alluvial vegetation units were included in the set of wetlands to be studied (*sensu* Mucina *et al.* 2006a) (See Figure 2.1); and
- within the mediterranean Western Coastal Slopes wetland region (SW.m: Cowan 1995) (See Figure 1.1);
- supralittoral, littoral and aquatic hydrological-habitats were all sampled in all of the wetlands where they were present; and

- the herbaceous structural vegetation unit was the focus of the present study, but scrub-shrub vegetation was sampled where encountered.

The study focused predominantly on isolated depression HGMs dominated by emergent plants within which the relationships between macrophytes and human stressors were investigated. Isolated depressions (SANBI 2009) were chosen to reduce the potentially homogenizing influences of the surface flow of water and associated nutrients between wetlands. A number of other wetland HGMs were sampled, namely flats, seeps, valley-bottom and floodplain wetlands where they were considered to hold the same or very similar habitat to the depressions. The intention of sampling multiple HGM types was to search for metrics with potential to be used in all of these wetland HGM types. An inventory of the abiotic details characterizing each wetland is presented in Appendix 2 and the biotic classification of each wetland is presented in Appendix 1.

2.2.4 Development of a Human Disturbance Score

An assessment method for integrating the different anthropogenic stressors, and thereby scoring the cumulative amount of disturbance impacting on each wetland, was formulated as part of this study. Assessment of the variety, intensity and extent of land-uses was used as an estimate of the degree of human disturbance affecting each wetland. This is referred to as the human disturbance score (HDS). The HDS of each wetland thus facilitated a ranking of the amount of stress that each wetland had been exposed to and an *a priori* categorization of wetlands as being '*minimally impaired*' and thus considered to be in a relatively natural and reference condition, relative to those impaired by anthropogenic disturbances. The term "reference" is used in this thesis to represent minimally impaired conditions. In keeping with the recommendation of Malan and Day (2005b), that in the absence of sufficient ecological understanding the graduation of ecological condition between relatively natural and highly impacted wetlands is difficult to determine, only reference and impaired categories of environmental condition were recognized.

This human disturbance assessment method was the product of several sources of information:

- The Western Cape Wetlands Inventory Datasheet (Dallas *et al.* 2006);
- The protocols stipulated by the Ohio Rapid Assessment Method, Version 5.0 (Mack 2001b);
- The Human Disturbance Score method of Gernes & Helgen (2002);

- The draft WET-Health protocol of Macfarlane *et al.* (2007 Draft, final version completed in 2008); as well as
- The draft Wetland Index of Habitat Integrity assessment tool (final version Rountree *et al.* 2007).

The tools and assessments that this process was adapted from all followed the procedure of determining an overall level of impact for each wetland being assessed. Whilst this procedure was followed it was realised that hydrological stressors impacting on the availability of water are not necessarily cumulative in the same direction. Water can be added or subtracted from a wetland by different anthropogenic influences. To determine the change in total water availability, impacts causing water loss must be scored negatively whilst those causing water gain would be scored positively. Both states of change qualify as disturbance, however, and from a cumulative disturbance perspective both count to increase disturbance, hence for ranking purposes both water gain and loss were scored positively so as to increase the human disturbance score.

The impact of human landuse activities were scored for the intensity and spatial extent of their impact on four major aspects of the wetland environment (the field sheet is shown in Appendix 3):

- water quality;
- hydrology;
- physical geomorphological structure; and
- buffer width of indigenous vegetation.

The likely impact of each anthropogenic stressor within the wetland and within a radius of 500 meters from the wetland edge was qualitatively assessed based on expert opinion and scored as described in A – C below. These scores were collated into an overall rating of cumulative disturbance to water quality, hydrology and physical-structure. A measure of the loss in extent of the buffer width of intact indigenous and/or altered vegetation surrounding the wetland was also estimated and included in the disturbance score as described in D & E below. The higher the score the greater was the level of disturbance. Qualitative assessments were made, within the wetland, and outside of the wetland within the first 100 metres from the wetland edge and within the next 400 metres as three separate spatial units, the scores from which were then collated into a score per wetland.

The procedural steps in this qualitative assessment of human disturbance were as follows (a worked example of this procedure is shown in Table 2.3):

A. For water quality, hydrology and physical structure the spatial extent (0%, 1-25%, 25-50%, 50-90%, >90%: scored 0 to 4) and intensity (least to most: scored 0 to 5) of various landuse activities were combined into a qualitative score of expected impact on a wetlands environmental condition. A range of expected activities and land-uses was provided as a guide, but other disturbances noticed in the field were also included.
B. The intensity and extent scores of each landuse activity were multiplied together to give a rating (0-20) of the amount of impact on water quality, hydrology and physical-structure;
C. The landuse activity impact ratings in B were then summed as three separate gradients of disturbance: water quality, hydrology and physical-structure.

Table 2.3: Extract from Human Disturbance Score sheet in Appendix 3, showing measurement of extent, intensity and resultant impact score of disturbance per landuse activity as separated, per gradient of disturbance.

	Within Wetland (e.g. Ken01)***						
		Water Quality		Hydrology		Physical Structure	
Landuse/Activity	Extent	Intensity	**Impact**	Intensity	**Impact**	Intensity	**Impact**
Infilling	2	1	2	3	6	2	4
Sewage disposal	1	3	3	2	2	0	0
Solid waste	3	3	9	1	3	4	12
Water Abstraction	1	2.5	2.5	3	3	2	2
Sum of Impacts: "within wetland"			**16.5**		**14**		**18**

***The same exercise was repeated for the 100m and the next 400m spatial units of assessment around each wetland

D. The loss or decrease in width of the buffer zone vegetation was scored in the following way (and as indicated in Table 2.4):
    i. On the four points of the compass (N, E, S, W) as four separate quarters, the width of natural indigenous vegetation and/or transformed vegetation was estimated according to seven categories (measured from broadest [least impacted] to narrowest [most impacted]: scored from 0 to 6);
    ii. The average of the four quarters was taken as the buffer width condition score for the wetland. The Narrowest and Worst state reflects the greatest loss of buffer width and is thus represented by the highest score.

Table 2.4: Extract from Human Disturbance Score sheet in Appendix 3, showing an example of the measurement of loss of buffer width. (e.g. for wetland Ken01)

Width of terrestrial vegetation buffer		(0) natural state	(1) natural buffer > 50 meters	(2) natural buffer 25 – 50	(3) Transformed vegetation 25 – 50 meters	(4) Trf'd 10 – 25 meters	(5) Transformed < 10 meters	(6) None
North side		**0**						
East side			**1**					
South side					**2**			
West side						**3**		
Score	**6 / 4**	**0**	**1**		**2**	**3**		

E. The score for buffer width was added with the scores generated per disturbance gradient (water quality, hydrology and landscape physical structure disturbance) in stage C, to give a human disturbance score (HDS) for the entire wetland (Table 2.5).

Table 2.5 Extract from Human Disturbance Score (HDS) sheet in Appendix 3, showing addition of disturbance gradient scores across the spatial units and buffer width.

Gradient:	Water Quality			Hydrology			Physical Structure			Buffer Width
	in wetland	100m	500m	in wetland	100m	500m	in wetland	100m	500m	-
Subtotal scores	16.5	X	Y	14	X	Y	18	X	Y	1.5
HDS =	(16.5	+ X	+ Y)	+ (14	+ X	+ Y)	+ (18	+ X	+ Y)	+ 1.5

*Spatial unit scores for 100m and 500m are marked X and Y respectively for each disturbance gradient.

Total scores were dependent on the number of land-use activities. Higher values thus represent greater disturbance on which no upper score limit was placed. The score sheet for each wetland and a summary page of all scores are presented in the CD of data appended to this thesis (Appendix 6).

A bar graph of the HDS for the sampled wetlands revealed no obvious separations that suggested category boundaries between reference and impaired wetlands (Figure 2.3). The data set was therefore divided into two categories, consisting of:

- seventeen reference wetlands with HDS scores ≤75 and ranging from 17 to 73; and
- forty two wetlands considered impaired by the degree of human impact with scores ranging from 77 to 251.

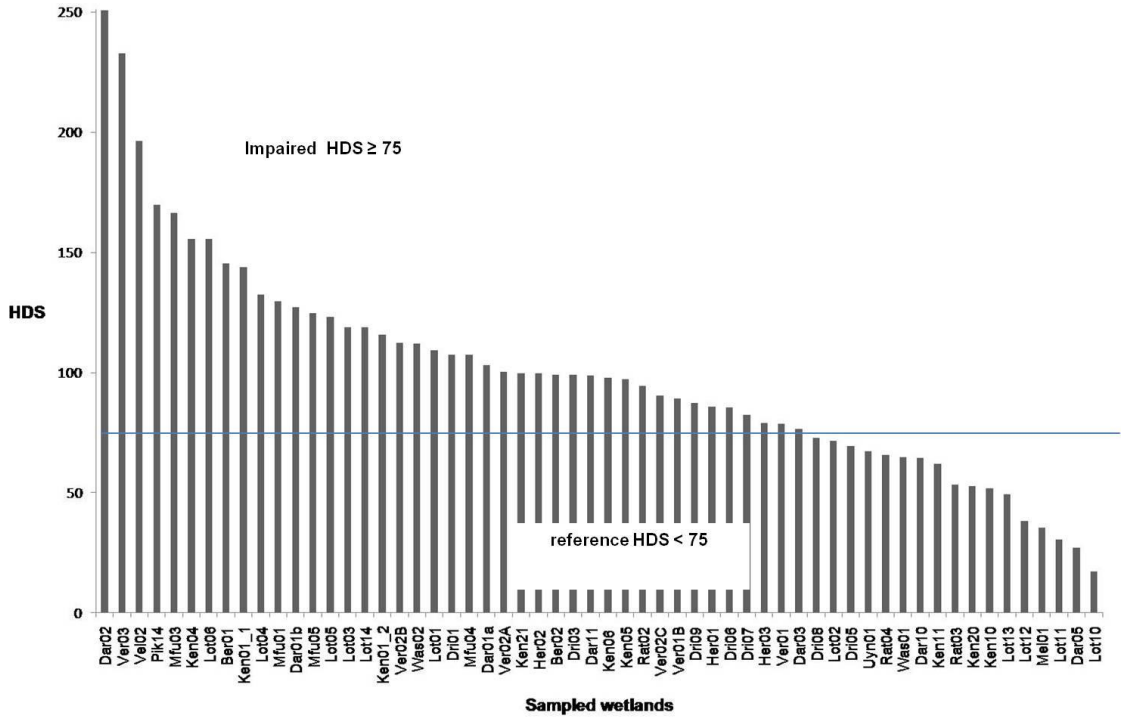


Figure 2.3: Chart of Human Disturbance Score (HDS) in each wetland sampled in the Western Coastal Slopes region of the Cape Coastal lowlands.

Best professional judgement of the relative degree of human impact on environmental condition was used to categorize those wetlands that fell close to the boundary between reference and impaired categories.

2.2.4.1 Nutrient load correlation with human disturbance

Nutrient concentrations in soil and water were also measured (as described in Section 2.2.5 & 2.2.6 below) and used as independent indicators of the influence of agriculture and urban land-uses. Eutrophic concentrations of any of these nutrients were considered to indicate human disturbance and thereby assist in the corroboration of categories of disturbance. For available phosphorus in the substrate of wetlands, values that were outliers from the wetland data set and that represented two standard deviations greater than the mean values reported for Fynbos and Strandveld (9.4 ± 2 and 60.8 ± 6 mg $P.kg^{-1}$) (Cowling & Holmes 1992, Witkowski & Mitchell 1987) were considered to represent eutrophic concentrations (Corry in press). Accordingly eutrophic phosphorous concentration in the substrate of Fynbos-associated wetland Ken20 (269 mg $P.kg^{-1}$) was used to reassign this wetland to the impaired category despite HDS value of less than 75.

2.2.5 Assessment of water samples: physical & chemical properties

i. Physical variables

Oxidation-reduction (redox) potential and pH were measured using a Crison pH25 meter (accurate to 1 mV and 0.01pH unit). Dissolved oxygen was measured using a Crison OXI45 oxygen meter (accurate to 0.01mg/L). Electrical conductivity was recorded using a Crison CM35 conductivity meter (accurate to 0.01µS/cm). Turbidity was measured from the water column immediately below the surface at two randomly selected points in each wetland using a Hach 2100P turbidimeter (accurate to 0.5 nephelometric turbidity units (NTU), the average of which was used for further analyses. Each of these meters was calibrated using appropriate standards and methods. Measurement of turbidity using NTU is a comparison of the intensity of light scattered by the sample whereby the greater the intensity of scattered light, the greater the turbidity (McCarthy *et al.* 1974, US EPA 1979). Precipitation of dissolved constituents (for example, iron) causes measured turbidity values to be high. Coloured solutes can cause measured turbidity values to be high; hence, tannins present in many Cape waters may lower NTU; however, the Hach 2100P turbidity meter's optical system compensates for colour in the sample, light fluctuation and stray light and hence in theory coloured solutes should not unduly increase NTU readings (Hach Website 2010).

ii. Nutrients

Two-litre surface water samples were collected from five locations within each wetland and pooled to form a bulk 10L sample, which was then thoroughly mixed and sub-sampled to obtain a 200mL sample for analysis of nutrients levels in the laboratory. Water quality analyses were carried out at the Department of Oceanography, University of Cape Town. Combined nitrate – and nitrite – nitrogen (NO_3 and NO_2 – N), and ammonium – nitrogen (NH_4 – N) concentrations were estimated using a Lachat Flow Injection Analyser, as follows: ammonium was measured using Lachat's QuikChem® Method 31-107-06-1, based on the Berthelot reaction in which indophenol blue is generated; nitrate and/or nitrite were estimated using Lachat's QuikChem® Method 31-107-04-1-E, in which nitrate is converted to nitrite and diazotized with sulfanilamide to form an azo dye. Approximate detection limits are 2.5µg.L^{-1} N for nitrate and nitrite and 5µg.L^{-1} N for ammonium. Details of the methods may be found at http://www.lachatinstruments.com. Total Inorganic Nitrogen (TIN) levels per wetland were determined from the addition of the concentrations of nitrate, nitrite and ammonium.

Inorganic phosphorus as ortho-phosphate (PO_4–P) was measured manually by forming an antimony-phospho-molybdate complex using the method of Murphy and Riley (1962) adapted to a 5ml sample size. The procedure is accurate for very low levels of phosphorus having been specifically designed to measure concentrations from 1 to 160µg.L^{-1} and is accurate to a maximum concentration of 2000µg.L^{-1} (Murphy & Riley 1962).

Some of the assessed wetlands had no standing water, being seeps or saturated flats, or were assessed at a stage when no surface water was present. The number of wetlands for which each physico-chemical parameter was measured is shown in Table 2.6.

Table 2.6: Water quality variables and number of wetlands for which measurements were taken. Total wetland count n = 59.

Water Variable	# of Wetlands
Redox (mV)	23
pH	50
Dissolved Oxygen mg.L^{-1}	32
Conductivity (mS.m^{-1})	50
Turbidity (NTU)	43
NH_4 (µg.L^{-1})	34
NO_3 + NO_2 (µg.L^{-1})	33
PO_4 (µg.L-1)	33

The values for these water variables for each wetland in which they were measured are presented in the appended CD (Appendix 6). Water variables were captured for most of the wetlands from the Cape Flats but logistical problems resulted in only limited water physico-chemical data being captured in the wetlands of the West Coast and Overberg regions.

2.2.6 Assessment of soil samples

In the first 16 wetlands sampled, only one or two soil samples were collected for analysis (n=20 samples). In the second phase of wetland sampling, soil samples were taken from almost every plot where vegetation was sampled (n = 242 samples).

Approximately one kilogram of soil was collected at each sampling point to ensure that, after drying, sufficient mass (250grams) would remain for analyses. All organic litter was removed from the soil surface and large pieces of organic matter were removed from each sample by hand. The soil was excavated and collected to a maximum depth of

25cm. Soil samples were stored in individual plastic bags, until they could be spread out and air-dried before being delivered for laboratory analysis.

2.2.6.1 Soil Sample Analysis

Soil particulate and chemical composition were analyzed in accordance with the recommendations of the Biological Assessment of Wetlands Working Group (US EPA 2002b; other than the determination of concentrations of the metals zinc, manganese, copper and boron which, were not performed). Soils were analyzed at BEMLAB (Pty) Ltd, Somerset West. Two blind duplicate samples were sent as a quality control to check the accuracy of analyses. Soils were again air dried overnight at BEMLAB before analyses were performed.

The following soil variables were determined:

- **Soil particle size: silt/clay/sand** distribution was determined by the mechanical hydrometer method (Van der Watt 1966).
- **Soil pH (KCl)** was determined by stirring 10g soil in 25mL 1M KCl at 180 r.p.m. for 5 seconds; after standing for 50 minutes the solution was re-stirred before measuring the supernatant pH with a calibrated meter.
- **Resistance (Ohm):** determination of electrical resistance was performed on a paste made from the soil by mixing with de-ionized water and using US Bureau of Soil Standards electrodes and a resistance bridge.
- **Bulk density (kg.m^{-3})** is the dry mass of the sample divided by the volume of the sample as determined by weighing a 50 mL volume of sieved soil (Hillel 1982, Jury *et al.* 1991).
- **Titratable acidity H^+ (cmol.kg^{-1}) (For soils with pH ≤ 6.1):** 5 grams of soil were shaken together with the extractant solution (potassium sulphate / potassium acetate, phenolphthalein and potassium hydroxide 0.1M), before filtering, and titrating with sodium hydroxide 0.05M (Eksteen 1969).
- **Total nitrogen (%):** determined by digestion with a FP-528 Nitrogen Analyzer (Leco Corporation, St Joseph, USA). Percent nitrogen was multiplied by 10 000 to convert to mg N.kg^{-1}. The mg N.kg^{-1} was divided by bulk density to determine the mg N.m^{-3}; this value was then divided by 1000 to convert to g N.m^{-3} and further divided by four to determine the g N.m^{-2} in the top 25 cm of the soil profile.
- **Phosphorus Bray No. 2 (mg.kg^{-1}) (For soils with pH < 6.9):** Plant available phosphorus was determined in soils with pH less than 6.9 with the Bray No. 2 reagent. The soil was prepared for Bray No. 2 P analysis by shaking 6.6 grams of soil in Bray 2 solution (150mL ammonium fluoride in 4L of water with 50mL of HCl) (Bray

& Kurtz 1945) before filtering and analyzing using an inductively coupled plasma opticam emission spectrometer (ICP-OES) (Varian Vista MPX, Melbourne, Australia).

- **Phosphorus Olsen (mg.kg^{-1}) (For soils with pH ≥ 6.9):** For soils with a **pH greater than or equal to 6.9**, the determination of plant available phosphorus was carried out using the Olsen reagent. These soils were prepared for analysis by shaking 5 grams of soil with sodium bicarbonate solution 0.5M (Olsen and Sommers 1982) before filtering and analyzing using an ICP-OES.
- **Potassium (mg.kg^{-1}):** exchangeable potassium, measured as described below, expressed in mg/kg. (($cmol_cK.kg^{-1}$) x 391.2 = mg $K.kg^{-1}$)
- **Percent organic matter (Walkley-Black) (For soils with pH ≥ 6.5):** determined by the Walkley Black method (Walkley & Black 1934).
- **Percent organic matter (LECO C/N-Analyzer) (For soils with pH < 6.5):** determined with a LECO CN-Analyser (Leco Corporation, St Joseph, USA).
- **Cation Exchange Capacity ($cmol_c.kg^{-1}$ or centimole of charge per kg):** was determined by following the procedure of Chapman (1965) in which 10 grams of soil are washed three times with 30mL of 0.2M ammonium acetate; then washed three times with a 1:1 water and methylated spirits mixture after which soil is eluted with 0.2M potassium sulphate and ammonium was measured using an auto-analyzer.
- **Exchangeable cations ($cmol_c.kg^{-1}$) calcium, magnesium, potassium, sodium:** were displaced from 10g soil with 25mL of 0.2M ammonium acetate. The samples were filtered through Reeve Angel Grade 307 filter paper, made up to 200mL and thereafter exchangeable cations of potassium (K), sodium (Na), calcium (Ca) and magnesium (Mg) were measured using ICP-OES analysis.
- **Water-soluble cations ($cmol_c.kg^{-1}$) of calcium, magnesium, potassium, sodium:** a saturated paste was prepared with de-ionised water, the soil and solution were separated by centrifugation and soluble cation concentration of the supernatant was determined by ICP-OES using appropriate standards.

Detection limits and calibration ranges of these analyses (as displayed in Table 2.7) are considered sufficient for the intents and purposes of the present study (Kotze pers. com.) and all measurements fell within bounds of the ranges provided here.

Table 2.7: Detection limits of soil nutrient analyses.

Analyte	Lower Limit of Detection	Lowest Quantifiable Concentration	Uncertainty of Measurement (%)	Calibration Range
P- Olsen	0.10 ($mg.kg^{-1}$)	0.32 ($mg.kg^{-1}$)	14.7	0-5 ($mg.kg^{-1}$)
P- Bray No. 2	0.14 ($mg.kg^{-1}$)	0.46 ($mg.kg^{-1}$)	3.4	0-10 ($mg.kg^{-1}$)
C% - Walkley-Black	0.01 $\%^m/_m$	0.05 $\%^m/_m$	14.6 $\%^m/_m$	0-12%
Resistance	-	-	15 %	10 - 10,000 (Ω)
N (Leco)	0.04 $\%^m/_m$	0.053 $\%^m/_m$	6.9 $\%^m/_m$	0 – 100 $\%^m/_m$
C (Leco)	0.01 $\%^m/_m$	0.04 $\%^m/_m$	11.3 $\%^m/_m$	0 – 100 $\%^m/_m$
Ca (ICP-OES)	N/A	N/A	3.9%	0 – 1.50 $\%^m/_m$***
K (ICP-OES)	N/A	N/A	3.7%	0 – 0.05 $\%^m/_m$
Mg (ICP-OES)	N/A	N/A	3.4%	0 – 0.80 $\%^m/_m$
Na (ICP-OES)	N/A	N/A	21.6%	0 – 0.08 $\%^m/_m$

(%m/m = the mass of analyte per mass of sample expressed as percentage)

*** A calibration range of 0 – 1.50 $\%^m/_m$ for Ca means a range of 0 to 15000$mg.kg^{-1}$ and is sufficient for current analysis purposes.

The values of the soil variables for each wetland and for each relevé in which they were recorded are presented in the appended CD (Appendix 6).

2.2.7 Other environmental variables

These variables were recorded in the field sheet for each vegetation sample (See Appendix 4) other than the climatic data, which was sourced independently as cited below and not measured in the field.

i. Climatic variables for each wetland

Climatic variables for each wetland were extracted from the "2002 (*a*)" data base of the South African Atlas of Climatology and Agrohydrology (Schulze 2006). The climate data are interpolated values at a resolution of 1' by 1' of a degree, derived from a network of recording stations using regression-type approaches (Schulze 2006; for method of interpolation, see Dent *et al.* 1987). The variables extracted were:

- Mean daily minimum Temperature (ºC);
- Mean daily maximum Temperature (ºC);
- Mean annual precipitation (mm); and
- Mean annual potential evaporation (mm).

This climatic data is recorded per wetland in the environmental data in the appended CD (Appendix 6).

ii. Quantitative variables measured at plot scale

The following quantitative and semi-quantitative variables were collated with the soil variables that were measured per vegetation plot:

- Slope and aspect were estimated by eye;
- Soil depth in ordinal classes (<0.2m, 0.2-1.5m, >1.5m) was estimated by hand augering to a depth of 1.5 metres.

iii. Qualitative or descriptive variables

In the vegetation sampling process (in section 2.2.8 below) a number of environmental parameters were described or assigned to various categories at every vegetation sample plot. These variables were descriptive of the following (as per Ewart-Smith *et al.* 2006):

- **habitat:** slope, flat, hypersaline flat, channel, microdepression, basin, aquatic, water column;
- **hydrogeomorphology:**
    - landform (basin, flat, channel, slope),
    - HGM type (channel, floodplain, unchannelled valley bottom, depression, flat).
- **hydrology:** water flow velocity (none, slow, fast);
- **sedimentation/erosion:** erosion, stasis, deposition (chemical, mineral, organic)
- **vegetation utilization:** none, mowed, grazed, harvested, overgrazed, excessively harvested.

iv. Hydrological variables at plot and wetland scales

For every wetland the following hydrological variables were recorded:

- Wetland size in hectares (ha) was determined relative to seven categories (<0.5, 0.5-1, 1-5, 5-10, 10-20, 20-50, >50);
- Maximum annual water depth in meters (m) was estimated according to four categories (<0.5, 0.5-1, 1-2, >2).

The above categories were each assigned a score (1-7) and (1-4) respectively. An ordinal concept of water **volume per wetland** was thereby developed by multiplying size by depth scores (i.e. wetland size: 3 x depth class: 2 = volume category 6). This facilitated a ranking of the relative volume of water each in wetland.

For every vegetation-sample-plot the following hydrological variables were recorded:

- **Current water depth** (millimetres) above (representing inundation) or below the ground surface (water table depth);
- Estimation of **potential maximum depth** of annual inundation (millimetres);
- **Current hydrological condition:** dry, moist, saturated or inundated; and

- **Hydrological regime:** temporary, seasonal or permanent.

The combination of current depth, estimated potential depth, current hydrological condition and hydrological regime were used along with the habitat description to determine whether a sample was aquatic, littoral or supralittoral. A concept of the relative hydroregime as a measure of the duration and seasonality of wetness of every sample plot was thus established

2.2.8 Vegetation sampling

Within the confines of each wetland the vegetation was examined to obtain a holistic overview and to determine homogeneous and 'representative' stands of vegetation (*sensu* Braun Blanquet 1928). Within each wetland, several representative vegetation stands could be recognized in the different hydrological zones and for each stand at least one vegetation relevé was recorded using the Braun Blanquet method (Westhoff & van der Maarel 1978). In equivalent wetland vegetation to that found in the Fynbos Biome, herbaceous vegetation plots of one to four square meters are recommended as effective for the capture of local alpha diversity (Westhoff & van der Maarel 1978, Peet *et al*. 1998, Sieben *et al*. 2004). Sample plots were laid out such that any variation within the plot was minimized – in other words if the unit being assessed was very narrow a 1 x 1 or 0.5 x 0.5m plot would be more appropriate than a 2 x 2m plot. The 2 x 2m plot size was most commonly used (99% of all quadrats). The standardized plot sampling sheet (Appendix 4) was based on that of Sieben (2003).

Homogeneous stands of vegetation were chosen that best characterized the various hydrological zones for the entire wetland as well as for different vegetation habitat units (sand vs. clay soils or impacted and un-impacted) within these zones. Within each chosen homogeneous stand a single sampling plot that was considered to be a characteristic representation of the vegetation was assessed, resulting in a relevé (or list) of species representative of that stand. The number of relevés per wetland was thus dependent on the number of different stands of homogeneous and representative vegetation and ranged from 3 to 16 samples per wetland (average of 6.7 ± 0.4). The cover and/or abundance data for every relevés and an average for every wetland are separately reported as the species data in the appended CD (Appendix 6).

Cover and abundance of each species were recorded for each sample plot in the Barkman *et al.* (1964) adjusted scale of Braun-Blanquet (1928) as shown in Table 2.8.

These values were adjusted to a representative median percentage to assist interpretability of statistical results.

Table 2.8: Cover and abundance values, representative codes, and median percentage values. (After Barkman, Doing & Segal (1964))

Cover	Abundance	Braun Blanquet code	Median % cover
<5%	1	R	1
<5%	2 - 10	+	2
<5%	11 - 100	1	3
<5%	>100	2m	4
5 - 12.5%	-	2a	8
12.5 - 25%	-	2b	18
25 - 50%	-	3	38
50 - 75%	-	4	68
75 - 100%	-	5	88

2.2.8.1 Specimen collection and Identification

All species recorded in the field were reported by their Latin binomial when identification was certain. Species that were unidentifiable in the field were collected for later identification. When possible, specimens were pressed in the field to ensure quality of preservation for identification and voucher specimen purposes (vouchers used to facilitate later field identifications). In cases where species were considered to be rare (fewer than 20 individuals of an unidentifiable species in existence at a site) specimens were photographed rather than collected to aid identification.

Specimens were identified in the Bolus Herbarium at the University of Cape Town. A number of difficult specimens were identified by experts in, or affiliated with, the Bolus Herbarium or the Compton Herbarium at Kirstenbosch. Non-vascular taxa were identified only to the lowest commonly recognizable taxonomic level. Nomenclature followed Goldblatt & Manning (2000), Germishuizen & Meyer (2003) and Govaerts *et al.* (2010). The full list of taxa recorded in the wetlands studied is reported in the appended CD (Appendix 6).

2.2.8.2 Wetland weighted-average species values

A cover/abundance value for each species per hydrological zone (supralittoral/littoral/aquatic) was calculated, providing an estimate of the representative cover or abundance for each species per zone or per wetland. The percentage area that each hydrological zone occupies in the wetland was used to weight the average value of

each species per hydrological zone. For instance species *F*'s average cover value from four different samples for the supralittoral zone in Wetland Dar01 = 48%. The supralittoral zone occupies 35% of wetland Dar01. Hence 0.35 x 48 suggests the supralittoral extent of species *F* has a cover value of 16.8% for wetland Dar01. Should *F* be found in other hydrological zones, the sum of the weighted cover values for each hydrological zone (i.e. $16.8\%_{supralittoral}$ + $x\%_{littoral}$) would equal the total cover value for species *F* for the wetland.

2.3 Methods of data analysis

Statistical difference between the different habitats within wetlands and between different *a priori* groups of wetlands as representative of different sub-regions, different bioregions (Rutherford *et al.* 2006), or different substrates (as determined by association with a given terrestrial vegetation unit (Rebelo *et al.* 2006)) were determined using analyses of similarity (ANOSIM), permutational analysis of variance (PERMANOVA) or discriminant analysis using canonical analysis of principal coordinates (CAP). Multivariate examination of the difference between impaired and minimally impaired sets of samples affords the use of the entire floristic assemblage and is considered more robust than searching simply for single indicator species (Fore 2003, Dahl 2004). Linear relationships of life-history groups (e.g. aliens or annuals), rather than single species, to disturbance levels were examined.

The distribution of samples in multi-dimensional space relates to how similar one sample is to another; be it a simple measure of geographical distance, or complex measure of floristic community data or of multiple environmental variables. Non-metric multidimensional scaling (nMDS) and cluster analysis are useful and complementary techniques that assist with the interpretation of the distribution of samples in multi-dimensional space (Clarke & Warwick 2001, Quinn & Keough 2003, Anderson *et al.* 2008). These techniques are methods of unconstrained ordination, in which, no imposition of *a priori* grouping is made on the samples and any groupings that emerge are a result of similarities between samples. Constrained ordination uses *a priori* grouping of samples and tests whether these groupings are accurate for all samples of a group.

Ordination by nMDS was used to display similarity of floristic assemblages (community structure) or of environmental similarity amongst sets of wetlands (Clarke & Warwick

2001). Constrained ordinations were performed using canonical analysis of principal coordinates (CAP) to discriminate between *a priori* determined groups.

Analysis of Similarity (ANOSIM) and Permutational Analysis of Variance (PERMANOVA) are means of quantifying the amount of difference between *a priori* determined groups. In combination with a general randomization approach to the generation of significance levels (Monte Carlo tests, *sensu* Hope 1968), the significance of the difference between groups is also ascertainable. As based upon the rank similarities between samples within the space of the resemblance matrix, the ANOSIM test statistic (*R*) reflects the observed differences *between samples in different* groups contrasted with differences among samples *within* groups (Clarke 1993, Clarke & Warwick 2001). Unlike the *F* statistic of analysis of variance (ANOVA), the *R* statistic of ANOSIM is an absolute measure of difference between groups and its value is thus directly comparable between different sets of *a priori* groups each with potentially different numbers of representative samples (Clarke 1993).

PERMANOVA tests the dissimilarity values generated by the 'sample by species' resemblance matrix on which permutations are based, through an analysis of variation of the estimates of pooled within-group variability, generating a test statistic of pseudo-*F* (or pseudo-*t*, for *a posteriori* pair-wise t-tests between subsets of groups within an analysis) (Anderson *et al.* 2008). PERMANOVA is akin to ANOVA in that the pseudo-*F* statistic is not comparable between analyses based on different numbers of samples.

ANOSIM and PERMANOVA are sensitive to differences in dispersion or homogeneity of variance (different ranges of variables, homoscedasticity) among groups (Clarke 1993 and Anderson *et al.* 2008). A test for homogeneity of dispersion, using the Permutational Dispersion (PERMDISP) routine can be performed to determine differences in dispersion of the variance between groups (Anderson 2006, Anderson *et al.* 2006). PERMDISP uses permutation of residuals (i.e., the permutation of samples among groups after centering all groups onto a common location) in order to generate *p*-values. PERMDISP detects differences in dispersion that, in many cases, are not substantial enough to inflate the error rates of the PERMANOVA or ANOSIM test. This is analogous with univariate analysis of variance (ANOVA) that is quite robust to many forms of heterogeneity of dispersion, especially in situations of large sample size (Box 1953).

The only assumptions about the data that are made in the use of ANOSIM or PERMANOVA are that samples are exchangeable under a true null hypothesis; in order

that exchanging the labels of the samples to generate significance levels can be performed (Clarke 1993, Anderson *et al.* 2008). This assumption is tantamount to assuming that the multivariate observations (samples) are *independent and identically distributed* under a true null hypothesis. For observational studies such as this one in which groups already occur naturally distributed in nature and we draw a (random) sample from them, we must assume exchangeability under a true null hypothesis (Kempthorne 1966). If the samples in different groups are not independent, for example if they are spatially correlated, or if their multivariate dispersion is not homogeneous, then they are not really exchangeable and randomly shuffling (permuting) said samples will destroy this inherent structure (Legendre 1993, Anderson *et al.* 2008). Results from such an invalidated analysis could lead to the incorrect acceptance that groups are consequentially and significantly different from one other. If nMDS ordination reveals differences between *a priori* groups then, partitioning each group such that samples from a group are not permutated with that of any other group, the generation of significance can be determined based only on within group variance.

The influence of environmental variability on floristic composition was explored with distance liner modelling (DistLM). This technique partitions the variation in data distribution according to a multiple regression model (based on the environmental variables), as selected by the user (e.g. forward, stepwise, best fit etc.). The "Best" procedure, examining the value of the selection criterion for all possible combinations of environmental variables, and the "AICc" selection criterion as adjusted for datasets in which the number of samples is not considerably larger than the number of environmental variables (Anderson *et al.* 2008) were used in this study. Environmental variables that are collinear at greater than 90% as determined in a search for multi-collinearity are removed before DistLM is performed (Anderson *et al.* 2008).

The statistical analysis package Plymouth Routines in Multivariate Ecological Research (PRIMER-E: Clarke & Warwick 2001, Clarke & Gorley 2006) and its add-on PERMANOVA (Anderson *et al.* 2008) were used for all analyses. Further explanations of statistical procedures are provided in the text where necessary.

CHAPTER 3

BIODIVERSITY PATTERNS AND PHYTOGEOGRAPHY

3.1 Introduction

Chapter 3 is an examination of the diversity and geographical distribution of wetland plants in the mediterranean Western Coastal Slopes wetland region of the Cape coastal lowlands of the Fynbos Biome. Species richness, uniqueness and field sampling effort are investigated with species estimators and species-accumulation and rarefaction curves. Comparison of floristic community structure is performed by using datasets of species cover/abundance and life-history groups. Species and life-history groups that are common to multiple wetlands throughout the region and are suggestive of similarity between wetlands are listed. Examination of the influence of hydrological zonation, as representative of habitat difference (See Section 1.5.2), is made by a comparison of the vegetation of supralittoral and littoral zones of all wetlands sampled. The potential for comparison of the floristic community structure of the supralittoral zone of wetlands from multiple HGMs but with otherwise naturally similar phytogeographical drivers is examined. Similarity between this habitat unit in multiple HGMs would reveal the applicability of supralittoral plants for phytoassessment of different wetland hydrogeomorphic types (See Section 1.5). Examination of the distribution of wetland vegetation at the biome, bioregion and mapped vegetation unit scales (*sensu* Section 1.6.1) tests the wetland vegetation azonality hypothesis (See Section 1.4). This examination of the geographical distribution of wetland vegetation also facilitates the identification of phytogeographic regions with relatively homogenous sets of wetland vegetation that result from limited natural environmental differences. Within such distinct phytogeographical units of wetland vegetation the effect of impairment (anthropogenic disturbance) on indigenous species distribution (homogenization) and cover /abundance is examined (*sensu* Section 1.2.3). This chapter thus explores whether wetlands in the mediterranean Western Coastal Slope region of the Cape coastal lowlands constitute multiple units or only a single homogenous unit of vegetation. This information is an essential step in the process of the development of phytoassessment.

3.2 Comments on the analytical methods employed

3.2.1 Species diversity

i. Richness

Field studies of species richness typically underestimate the number of species that exist within a habitat. The difficulty of detecting all species or of accurately calculating species cover or abundance within a sample (e.g. Kent & Coker 1992, Magurran 1994), has led to the use of a range of species richness estimators and of species rarefaction curves to ascertain the degree of sampling representivity (Gotelli & Colwell 2001). Based on the incidence of species inventoried to occur within a subset of an assessed habitat, these estimators and graphical techniques predict an approximation of total species richness and whether the subset sampled is representative (Soberón & Llorente 1993, Gotelli & Colwell 2001, Colwell 2009). The more species that are unique to a sample (wetland), the more species are predicted to be present in the sampling 'universe' that remain to be captured during a survey. Graphed curves of species accumulation (observed species) and rarefaction (estimates) approach an asymptote when the number of samples (wetlands) is an adequate representation of the habitat under study (e.g. Colwell & Coddington 1994). Species richness estimation indexes and species accumulation and rarefaction curves were generated for the whole study set of the wetlands of the Cape coastal lowlands (n=59 wetlands) and for the western, central and eastern sub-regions, the West Coast, (n=16), the Cape Flats (n=33) and the Overberg (n=10). The incidence-based Chao 2, and the first-order Jackknife estimators predict the number of unseen species and thus overall species richness based on the number of rare species, specifically those observed only in one wetland (unique species) (Colwell & Coddington 1994). The second-order Jackknife estimator predicts unseen species and thus richness based on species observed to occur in only one or in exactly two wetlands, thereby reducing the emphasis of uniques and duplciates. The bootstrap indicator predicts richness based on the proportion of wetlands in a dataset that contain each species (e.g.: 23 of the 59 wetlands contain species x, whilst only 11 contain species y), thereby focusing on proportional similarity as opposed to difference. Each of these species estimators therefore emphasizes frequency of occurrence of every species in the data set and estimates the possibility of having missed species during the inventory process (sampling) based on the number of species that are rare within the whole or portions of the data set. Each estimator thus provides a level of information that the other curves do not. Used in combination these curves provide a relatively robust method of estimating

total species richness relative to the species observed (S_{obs}) or inventoried during the sampling process (Chao *et al.* 2005).

The parametric Michaelis-Menten model of species richness was also calculated to predict the number of wetlands required to sample 50% of estimated total species richness (S_{max}) for any given sample set. These richness estimators (other than Michaelis-Menten which consistently underestimates S_{obs} (Clarke & Warwick 2001)) were plotted as rarefaction curves against the accumulation of observed species (S_{obs}). Species richness estimators and incidence coverage estimators were calculated using Estimate S (Colwell 2009). Species accumulation and rarefaction curves were produced using PRIMER-E (Clarke & Warwick 2001).

ii. Diversity and Similarity

The definition of beta diversity has recently been clarified to mean the variation in the identities of species among sites; and being either directional turnover along a gradient or non-directional variation (Anderson *et al.* 2011). Beta diversity, thus defined, provides a direct link between biodiversity at local scales, namely the alpha diversity within samples (e.g. wetlands), and the broader regional or biome-wide species pool, namely gamma diversity (Whittaker 1960 & 1972). Total species richness of wetlands in the Cape coastal lowlands represents gamma diversity, the richness of each sub-region represents the gamma diversity of each sub-region and difference between these sub-region values represents the beta diversity or variation in species among different sub-regions.

Similarity of floristic community structure (based on species incidence and cover/abundance) within and between sub-regions and within the Cape coastal lowlands was assessed using a number of similarity indices that provide a measure of diversity. The Bray-Curtis index of similarity measures the ratio of the variation of species between sites (or turnover along a gradient between sites) to the total species richness of compared sites (Legendre & Legendre 2003). The Jaccard and Sørensen abundance-based similarity indices were also used to assess community similarity. These two indices are adjusted for unseen species, take into account large numbers of unique species and accommodate differing sample sizes and undersampled populations (Chao *et al.* 2005) Percentage cover was used in place of numerical abundance for these indices as is typically performed in studies of vegetation (Magurran 2004, Chao *et al.* 2005, Colwell 2009). Estimate S (Colwell 2009) was used to determine the values of

these similarity indices. The value of these similarity indices, as generated by Estimate S, is bound between 0 and 1, where 1 means the compared sites have the same composition (that is, they share all the species), and 0, meaning that the compared sites do not share any species (Colwell 2009). This software package relies on the use of integers and hence any species that had cover of less than 0.5 % within a wetland were not included in the determination of community similarity. This thereby excludes species that have limited cover (those species with small growth forms and limited abundance) from the calculation of beta diversity differences among sites. The diversity differences generated, thus, must be considered as conservative estimates of total diversity.

3.2.2. Hydrological zones

As mentioned in Section 1.5, turnover or alteration of vegetation along a moisture gradient (i.e. hydrological zonation) within a wetland (i.e.: at the landscape scale) has been more extensively researched in the southern African context (Kotze *et al.* 1994 and 1996, Kotze & O'Connor 2000, Ellery *et al.* 2003, Sieben *et al.* 2004 and 2010) than the effects of other environmental determinants on wetland species distribution at the regional scale. Within wetlands, different hydrological zones exist that are suggestive of habitats that may support different species (Section 1.5.2, and US EPA 2002b). For the purposes of the present study a characterization of the vegetation of different habitat units within each wetland was sought and an *a priori* characterization of hydrological zones meant that each separate vegetation plot sampled in the present study was assigned to a hydrological zone with the null hypothesis that each zone represents a different vegetation community. This contrasts with the sampling of vegetation along the hydrological gradient in order to determine where species occur on this gradient as in the studies of Kotze *et al.* (1994, 1996), Kotze & O'Connor (2000) and Ellery *et al.* (2003). As samples of each hydrological zone represent different hydrological categories and not a linear moisture gradient along a transect, beta diversity differences between the supralittoral and littoral (hydrological habitats) must be tested as variation between categories (rather than turnover along the hydrological gradient). The 396 vegetation samples recorded in the present study constitute too great a number to facilitate the use of ordination techniques to search for variation in community structure and beta diversity between hydrological categories (Clarke & Warwick 2001). Analysis of similarity (ANOSIM) was thus used to test for differences between the community structure of different hydrological zones.

3.2.3 Azonality of wetland phytogeography

Azonal distribution of the wetland Cape Lowland Freshwater vegetation unit (Mucina *et al.* 2006a) would suggest that limited difference be discernable between the vegetation contained within wetlands in areas with different climatic influences and associated with different terrestrial vegetation units across the mediterranean Western Coastal Slopes region (Cowan 1995) of the Cape coastal lowlands. Examination of the resemblance of the assemblage of species (community structure) found in each wetland relative to other wetlands can be performed by the projection of the multidimensional resemblance data onto two dimensions. Such a projection can be used to identify discontinuities (or outliers) in multivariate data sets and thus reveal distributional disjunctions (Clarke & Warwick 2001, Quinn & Keough 2003, Anderson *et al.* 2008). Unconstrained ordination using non-metric multi-dimensional scaling and constrained ordination using discriminant analysis are both visual techniques that assist with the interpretation of the distribution of samples in multi-dimensional space. Disjunctions in distribution of wetlands in the multidimensional resemblance space would suggest differences in their community structure. Such differences in community structure may be caused by a multitude of drivers including natural or unnatural determinants of the spatial distribution of species. Comparison of the community structure of wetlands of each sub-region of the Cape coastal lowlands (western, central and eastern), of each bioregion (climatic difference), and associated with each terrestrial vegetation unit (edaphic difference) was performed using these ordinations techniques. Analysis of Similarity (ANOSIM) between *a priori* defined groups was also used to confirm the differences that are apparent in the ordinations. These analyses were performed with PRIMER-E (Clarke & Warwick 2001, Clarke & Gorley 2006).

i. Sub-regions

Wetlands were sampled on the West Coast, the Cape Flats and the Overberg as representative of areas in which considerable concentration of wetlands exist within the Cape coastal lowlands. As a first step toward examining the supposed azonality and broad distribution of lowland freshwater wetland vegetation the differences in floristic community structure between all wetlands was examined with unconstrained and constrained ordinations and with cluster analysis. Should considerable disjunctions occur in the floristic community structure of wetland vegetation between sub-regions this may contradict the supposed azonality of wetland vegetation.

ii. Bioregions

Climatic differences were used to delimit the extent of bioregions of terrestrial vegetation in the Fynbos biome thereby grouping terrestrial vegetation units that were already grouped by association with a geological substrate (e.g.: acidic sands *vs* calcareous sands) (Rutherford *et al.* 2006). The zonal bioregions of the "*South-west Fynbos"* and "*West Strandveld"* extend across western and central sub-regions of the Cape coastal lowlands, with the *South-west Fynbos* extending beyond this into the Overberg or eastern section of the Western Coastal Slopes region of the Cape coastal lowlands. An examination of the uniformity of the wetland vegetation in the different bioregions across the Fynbos biome can reveal whether macroclimatic and geological drivers influence distribution of wetland vegetation species. A comparison of the differences in floristic community structure between bioregions may reveal spatial differences that relate to climatic or geological constraints. Should floristic composition of wetlands from a given bioregion reflect greater affiliation to each other than to wetlands from different bioregions, this would further contradict the supposed azonality of wetland vegetation and suggest that, within the within the lowlands of the Fynbos biome, climatic and geological differences do influence distribution and composition of wetland vegetation.

Given the known climatic gradient of moisture availability with increasing summer aridity and increasing concentration of rainfall in the winter months from the south-east to the north-west of the Cape coastal lowlands (Rebelo *et al.* 2006, Schulze 2006, see Section 2.2) it is possible that a considerable climatic gradient is incorporated within each bioregion. Examination of differences in wetland vegetation within the *South-west Fynbos* bioregion, which contains the greatest north-west to south-east spread of all bioregions within the Cape coastal lowlands, may reveal variation that correlates with this climatic gradient of moisture availability. The influence of environmental variability on floristic composition between bioregions was explored with distance liner modelling (DistLM).

iii. National Freshwater Ecosystem Priority Areas

The National Freshwater Ecosystem Priority Areas (NFEPA) group wetlands within a regional context as based on similar geological and climatic conditions (Roux *et al.* 2006, see Section 1.4.3). In the Cape coastal lowlands this resulted in combining wetlands associated with terrestrial vegetation types (*sensu* Rebelo *et al.* 2006), which had similar geological substrates such as sand *vs* sandstone *vs* shale *vs* granite; as well as into units

with climatic similarity as based on the terrestrial bioregions of Rutherford *et al.* (2006). The NFEPA groups of wetland vegetation such as the "*South-west Sand Fynbos"* incorporate wetlands from an array of terrestrial units of vegetation on acidic sands that support Fynbos. This *South-west Sand Fynbos* NFEPA vegetation group differs from the South-west Fynbos Bioregion that also incorporate wetlands associated with sandstone Fynbos. The *South-west Sand Fynbos* group incorporates wetlands from the West Coast and the Cape Flats sub-regions into a single unit which is expected to hold similar vegetation that is different from that of the *"North-west Sand Fynbos"* vegetation group on the West Coast. Difference in the floristic community structure of wetlands in the *South-west* and *North-west Sand Fynbos* is tested using ANOSIM along with an exploration of the differences of each of the sets of wetlands representative of the different units of terrestrial vegetation within and between each of these NFEPA vegetation groups.

iv. Zonal interface – terrestrial vegetation units

The influence of geological substrate and climate on wetland vegetation distribution is explored across the whole of the Cape coastal lowlands by comparing species composition of wetlands associated with different terrestrial vegetation units. The terrestrial vegetation units are considered to be distributed based on zonal influences of macroclimatic constraints (Rebelo *et al.* 2006). Essentially though, I consider these terrestrial vegetation units as intrazonal units within the Fynbos biome of the global mediterranean zonobiome. For the purposes of this research the units of wetland vegetation sampled exclusively within a terrestrial vegetation unit are thus each considered to represent potentially different intrazonal units of vegetation. Within ordinations, should separation be evident between the wetlands associated with each of these terrestrial vegetation units, then the distribution of wetland vegetation must be considered to be constrained by similar parameters affecting the distribution of terrestrial vegetation units.

3.2.4 Biotic exchange, species homogenization and cover

Impairment of wetland ecosystem condition by human-induced disturbances has been shown to alter plant species assemblages (Simon *et al.* 2001, Gernes & Helgen 2002, Mack 2007). Alien species invasion alters community structure reducing the numerical diversity of indigenous species and the area that native/indigenous species are able to occupy in the process called biotic exchange (e.g. Sala *et al.* 2000, Slobodkin 2001, Collins *et al.* 2002, Rouget *et al.* 2003; See Section 1.2.1, conversely see also Davis

2003 and Houlahan & Findlay 2004, Section 1.2.2). Biotic exchange causes, over time, a shift toward a more homogenous species pool as a result of alien invaders that displace sensitive native species (Wilcox *et al.* 2002, Gernes & Helgen 2002, Fore 2003). Hence, whilst reference wetlands of different vegetation types may be floristically different from one another, disturbances such as biotic exchange and the resultant homogenization of communities may result in impaired wetlands with floristic assemblages that are no longer significantly different between the vegetation types. The number of alien species and the extent of their cover relative to indigenous species may also be expected to be higher in wetlands that are more impacted by human disturbance (Planty-Tabacchi *et al.* 1996) and biotic exchange than those that are least impaired by these impacts.

If biotic exchange reduces the floristic differences between units of wetland vegetation, then in an ordination of the floristic resemblance of numerous types of wetland vegetation the impaired wetlands would reduce the separation between different vegetation units. This would suggest that disturbance homogenizes overall assemblage resemblance. This hypothesis was tested using the Cape Flats data set.

3.2.5 Supralittoral vegetation of HGMs

Hydrogeomorphic units (HGMs) are expected to drive the development of different water column habitat types (SANBI 2009). In Chapter 1 it was hypothesized that the supralittoral zone from different HGMs may constitute a similar habitat unit that would support similar vegetation (Section 1.5). The potential to ascertain whether different HGM types hold similar floristic communities in their supralittoral zones can only be explored, using the current data set, if:

- edaphic and climatic drivers do not affect the distribution of wetland vegetation and it is azonal in the sense that was hypothesized by Walter 1973 and mapped by Mucina *et al.* (2006a); or if
- Sufficient wetlands from different HGMs but within the same phytogeographical area were sampled.

3.3 Results

The 373 species inventoried in 396 relevés are presented in Appendix 5 along with a descriptive growth form, and an indication of whether the species is annual, or perennial, indigenous or alien and the consistency of its affiliation with wetlands conditions (*sensu* Reed 1988). Any species and or genera that are considered alien within the Fynbos

biome were marked with an asterisk in all tables or text within which alien species are named and in the appended data CD (Appendix 6) . The incidence and cover/abundance of species in each sample are presented in Appendix 6. Of the 373 plant species, 352 were identified to species level with the remaining 21 taxa identified to genus level. Almost half (n = 177) of all taxa recorded occurred in only one wetland. Many invasive alien species (n = 73) were recorded with over 50% of these being widespread and only 29, occurring in only one wetland (uniques). Of the indigenous macrophyte species observed in vegetation samples, 151 occurred in more than one wetland. In these Cape coastal lowland wetlands 50 indigenous plant families and 102 genera occurred in more than one wetland. Of these, the Cyperaceae, Asteraceae and Poaceae were represented by 28, 19 and 19 genera respectively whilst the Iridaceae, Restionaceae, and Juncaceae were represented by 7, 6 and 5 genera. The indigenous species that were commonest in the wetlands of the study, as determined by largest cover values, and / or presence in the greatest number wetlands, are presented in Table 3.1.

Table 3.1: Indigenous species that were most common in the wetlands of the Cape coastal lowlands as chosen from species with the largest cover values and or presence in the greatest number of wetlands. The wetland or terrestrial association of each species (*sensu* Reed 1988; see table 1.2) was taken from Glen (unpublished – Appendix 6) or Goldblatt & Manning (2000). Life-history groups (*sensu* Galatowitsch *et al.* 2000) were developed from a combination of longevity (perennial/annual), origin and growth form.

Plants	Sub-regions and / or associated vegetation type	Perennial / Annual †	wetland association **	Growth form	Life History Groups (*sensu* Galatowitsch *et al.* 2000)***	Growth Forms (*sensu* Mucina *et al.* 2006b)
Amaranthaceae						
Sarcocornia cf. natalensis	WC & CF & Ob	p	fw+	herbaceous	IHP (i.e.Indig's Herb's Pere'l)	succulent herb
Aponogetonaceae						
Aponogeton distachyos	WC & Ob	p	Ow	herbaceous	IHP	aquatic herb
Araliaceae						
Centella asiatica	CF & Ob	p	Ow	herbaceous	IHP	Herb
Asteraceae						
Cotula coronopifolia	WC & Ob	a	Ow	herbaceous	IHAn (i.e.Ind's Herb's Ann'l)	succulent herb
Cotula turbinata	WC & CF & Ob	a	Fw	herbaceous	IHAn	Herb
Cotula vulgaris	WC & Ob	a	Ow	herbaceous	IHAn	Herb
Nidorella foetida	CF Strandveld	p	fw+	Woody	IW	low shrub
Plecostachys serpyllifolia	CF & Ob	p	fw+	herbaceous	IHP	Herb
Senecio halimifolius	CF & Ob (Hermanus)	p	fw+	Woody	IW	low shrub
Characeae †††						
Chara ecklonii	WC & CF	a	Ow	Algae	Ialgae	Macroalga
Crassulaceae						
Crassula glomerata	WC & CF & Ob	a	Fw	herbaceous	IHAn	succulent herb
Crassula natans	WC & CF & Ob	fa	Ow	herbaceous	IHAn	succulent herb
Cyperaceae						
Bolboschoenus maritimus	WC & CF & Ob	p	Ow	graminoid	IGP	Graminoid
Cladium mariscus	CF Strandveld	p	Ow	graminoid	IGP	mega-graminoid
Cyperus sphaerospermus	CF	p	Fw	graminoid	IGP	Graminoid
Cyperus textilis	WC & CF & Ob	p	fw+	graminoid	IGP	Graminoid
Eleocharis limosa	WC & Ob	p	Ow	graminoid	IGP	Graminoid
Ficinia nodosa	CF Strandveld	p	Fw	graminoid	IGP	Graminoid
Fuirena hirsuta	CF & Ob Fynbos	p	Ow	graminoid	IGP	Graminoid
Isolepis cernua	WC & CF & Ob	a	fw+	graminoid	IGAn	Graminoid
Isolepis hystrix	CF Fynbos	a	Ow	graminoid	IGAn	Graminoid
Isolepis rubicunda	CF & Ob	p	Ow	graminoid	IGP	Graminoid
Schoenoplectus cf. roylei	WC	a	Ow	graminoid	IGAn	Graminoid
Schoenoplectus scirpoideus	WC & Ob Saline	p	Ow	graminoid	IGP	mega-graminoid

Schoenus nigricans	CF Strandveld	p	Ow	graminoid	IGP	Graminoid
Scirpoides thunbergii	WC & OB	p	fw+	graminoid	IGP	Graminoid
Iridaceae						
Watsonia meriana	CF Fynbos	p	fw+	herbaceous	IHP	geophytic herb
Juncaceae						
Juncus capensis	CF & Ob	p	Fw	graminoid	IGP	Graminoid
Juncus kraussii	WC & CF & Ob	p	O	graminoid	IGP	Graminoid
Juncus oxycarpus	CF Fynbos	p	Ow	graminoid	IGP	Graminoid
Poaceae						
Cynodon dactylon	WC & CF & Ob	p	F	graminoid	IGP	Graminoid
Ehrharta calycina	WC & CF	fa-p	F	graminoid	IGAn	Graminoid
Eragrostis plana	WC & CF & Ob	p	Fw	graminoid	IGP	Graminoid
Imperata cylindrical	CF Strandveld	p	Ow	graminoid	IGP	Graminoid
Pennisetum macrourum	WC & CF & Ob	p	Ow	graminoid	IGP	mega-graminoid
Phragmites australis	WC & Ob	p	Ow	graminoid	IGP	mega-graminoid
Sporobolus virginicus	WC & CF & Ob	p	fw+	graminoid	IGP	Graminoid
Stenotaphrum secundatum	CF & OB	p	fw+	graminoid	IGP	Graminoid
Polygonaceae						
Persicaria decipiens	WC & CF & Ob	a	Ow	herbaceous	IHAn	Herb
Potamogetonaceae						
Potamogeton pectinatus	CF & Ob	a	Ow	herbaceous	IHAn	aquatic herb
Restionaceae						
Elegia nuda	CF & Ob	p	Ow	graminoid	IGP	Graminoid
Elegia tectorum	WC & CF & Ob	p	fw+	graminoid	IGP	Graminoid
Typhaceae						
Typha capensis	WC & CF & Ob	p	Ow	graminoid	IGP	mega-graminoid

† a= annual, p = perennial, fa = facultative annual, fa-p = facultative annual to perennial (After Goldblatt & Manning 2000)

***I = indigenous, G = Graminoid, H = herbaceous, W = woody, P = Perennial, An = annual

** A + sign indicates strong conviction of status: **(See Table 1.2)**

††† As the Characeae represented here were macroalgae that were visible with the naked eye and of large growth form, they were considered as macrophytic taxa and were included in the study.

3.3.1 Biotic diversity

i. Species Diversity

A summary of the number of species recorded from characteristic stands of vegetation in the 59 wetlands of the study and other summary values including the estimated species richness within the Cape coastal lowlands and each of its sub-regions are presented in Table 3.2. The Michaelis-Menten index suggested 30 wetlands would have been sufficient to sample 50% of the estimated species richness of the Cape coastal lowlands, whilst 13, 15 and 18 wetlands would, respectively, have been required to do so in the West Coast, Cape Flats and Overberg wetlands (Table 3.2 part A).

All wetlands supported a high percentage of uniques, (species occurring in only one wetland); ranging from approximately 45% in the entire Cape coastal lowlands and Cape Flats wetland data sets, to nearly 60% and 70% in the West Coast and Overberg wetlands respectively (Table 3.2, Part B). Thus a considerable number of species were unique to each sub-region: 17% to the West Coast, 54% to the Cape Flats and 68% to the Overberg. This high percentage of unique species is suggestive of considerable species turnover between sub-regions and resultant beta diversity differences between the sub-regions of the Cape coastal lowlands. Due to the high percentage unique species, low levels of floristic similarity were recorded between all wetlands within the Cape coastal lowlands as determined using the Bray-Curtis index of similarity (Table 3.2 part C). The use of cover and integer values in the generation of index values lead to the exclusion, on average, of 23 ± 2% percent of all species surveyed per wetland which occurred with average cover values of less than 0.5%. The exclusion of this number of species may either reduce similarity or increase similarity. Determining the Bray-Curtis index values using incidence data, thereby including all observed species, suggested only marginally more similarity or greater index values than deduced when species with mean wetland cover of less than 0.5% were excluded. This points to the facts that some of these species with <0.5% cover (e.g. *Isolepis cernua, Crassula glomerata, C. natans*) are widespread in the Cape coastal lowlands. The adjusted Jaccard and the Sørensen similarity indices (Chao *et al.* 2005) returned similarly low levels of inter-wetland similarity between all wetlands in the Cape coastal lowlands and within sub-regions. The various indices of similarity all returned values below 0.2, thus confirming that there is considerable variation in species between wetlands of the Cape coastal lowlands and within and between the macrophyte communities of each sub-region.

Only 27 observed species (7%) were shared between all three sub-regions of the Cape coastal lowlands suggesting that wetland species distribution is in general not broad within the Cape coastal lowlands. The 27 species shared by all of the sub-regions are listed in Table 3.3. Of these 27 shared species three alien (non-indigenous) and two indigenous species that are more typically associated with dryland or terrestrial conditions were observed along with a single alien wetland species whilst the remaining 21 shared species were indigenous wetland taxa. In Table 3.2 (part D), the term "terrestrial" refers to species that were recorded in the wetlands but that are more typically associated with dryland (terrestrial) than wetland conditions. Differentiation between terrestrial *vs* wetland taxa was determined according to the US Fish and Wildlife Service indicator categories (Reed 1988 – see Table 1.2) as used by Glen to assign such categories to Southern African macrophytes based on herbarium data (Glen unpublished in Appendix 6). Where a species was not included within Glen's (unpublished) list the habitat description provided by Goldblatt and Manning (2000) was used to decide terrestrial *vs* wetland status (This data is also presented in Appendix 6).

The number of macrophyte species observed in wetlands of the Cape coastal lowlands was between 60 and 80% of the estimated species richness (Table 3.2 part E) depending on the richness index used. The Cape Flats wetlands held a similar observed to estimated ratio of species richness as the whole Cape coastal lowlands. In the West Coast wetlands, the number of species observed was approximately 50 to 80% of estimated richness, whilst for the Overberg wetlands observed richness dropped to 40 to 80% of estimated richness. The bootstrap estimate of species richness, based on proportions of wetlands that contain each species, consistently yielded values closest to the observed richness as is evident in all of the graphs of species accumulation and rarefaction curves in figures 3.1 to 3.4. None of the other richness estimators yielded consistent results other than being consistently higher than the observed species richness.

Table 3.2: Species richness estimates and other summary values for macrophyte species sampled in the wetlands of the Cape coastal lowlands and the West Coast, Cape Flats and Overberg sub-regions thereof. Each richness estimate represents the mean (± SD for some estimators) for 50 randomizations of the sample order.

mediterranean Western Coastal Slope Region		Sub-regions		
	Cape coastal lowlands	West Coast	Cape Flats	Overberg
(A) Number of Wetlands				
No. of wetlands	59	16	33	10
No. wetlands holding 50% estimated species richness	30	13	15	18
(B) Number of species observed				
No. of species (*N*) S_{obs}	373	121	202	191
No. of species in only 1 wetland (uniques)	177	69	93	132
No. of species in only 2 wetlands	76	21	41	41
No. observed species unique to a sub-region	-	21	109	90
(C) Estimated species similarity – mean values (a shared species is one that occurs in more than one wetland)				
Mean no. shared species per wetland (Observed)	1.5	2	2.6	1.4
Mean no. shared species per wetland (Chao Estimate)	1.5	1.7	2.7	2.1
Bray-Curtis species similarity (mean)	0.09	0.14	0.15	0.07
adjusted Jaccard wetland similarity****	0.09 (± 0.07)	0.13 (± 0.09)	0.17 (± 0)	0.07 (± 0)
adjusted Sørensen similarity***	0.15 (± 0.1)	0.21 (± 0.12)	0.26 (± 0)	0.12 (± 0)
(D) Life history differences				
††No. species Terrestrial *vs* Wetland	104 *vs* 270	34 *vs* 89	53 *vs* 148	43 *vs* 148
No. species Alien *vs* Indigenous	74 *vs* 300	37 *vs* 86	44 *vs* 158	24 *vs* 167
No. species Annual *vs* Perennial	96 *vs* 278	45 *vs* 77	48 *vs* 153	35 *vs* 156
(E) Estimated species richness indexes				
Chao 2 (±SD)	583 (±42)	230 (±36)	309 (±29)	408 (±52)
Jack 1	549 (±31)	187 (±16)	293 (±19)	312 (±29)
Jack 2	649	229	343	389
Bootstrap	451	150	243	243
S_{max}**	530	208	280	554

††Terrestrial / Wetland as per frequency of occurrence in dryland / wetland habitat (Glen unpublished & Goldblatt & Manning 2000).

****Chao's Jaccard Abundance-based similarity index adjusted for unseen species (Chao *et al.* 2005)

**Chao's Sørensen Abundance-based similarity index adjusted for unseen species (Chao *et al.* 2005)

**Smax: estimated total species richness (Michaelis-Menten in Clarke & Warwick 2001)

The species accumulation plot and the rarefaction curves of the species richness estimators did not reach an obvious asymptote, although, the rarefaction curve of the Chao 2 richness estimator was close to levelling off in all of the sampling regions bar the West Coast (compare Figures 3.2 *vs* Figures 3.1, 3.3 and 3.4), which had the lowest number of uniques of any of the sub-regions. Rarefaction (comparison) between the first few samples of a data set with a large number of uniques causes Chao 2 to over-estimate species richness at the beginning of the sample set (Williams *et al.* 2007) as is evident by the spike in richness at the start of this curve in each of Figures 3.1 to 3.4. The order in which wetlands are accumulated was randomized by 999 permutations in all

of these analyses. Whilst the rarefaction curve for Chao 2 levelled off for all areas other than the West Coast, due to the considerable amount of biotic distinctness within these data sets, second-order Jackknife and bootstrap estimators, which are both less dependent on unique species, may present more realistic estimates. Using these latter two estimators and the first-order Jackknife, the rate of accumulation of new species with increased sampling effort did not level off enough for sampling effort to be considered satisfactory in the West Coast or Overberg sub-regions. The Cape Flats was the most comprehensively sampled sub-region and shows the closest approach to asymptotes for all indexes that were graphed (Figure 3.3). Of all wetlands in the Cape coastal lowlands, the wetlands of the Cape Flats were most comprehensively sampled and represent an adequate representation of the habitat. These Cape Flats wetlands thus present the best data set within which to examine the impacts of climate and edaphic properties as well as disturbance on species distribution.

Table 3.3: Indigenous and alien species that occurred in wetlands of all sub-regions of the Cape coastal lowlands. The wetland or terrestrial association of each species (*sensu* Reed 1988; see table 1.2) was taken from Glen (unpublished) or Goldblatt & Manning (2000). Life history groups (*sensu* Galatowitsch *et al.* 2000) were developed from a combination of longevity (perennial/annual), origin and growth form.

Plants	Perennial / Annual †	wetland association (see table 1.2)**	Origin	Growth form	Life History Groups ***	Growth Forms (*sensu* Mucina *et al.* 2006b)
Aizoaceae						
Carpobrotus edulis	p	F	Indigenous	Herbaceous	IHP	succulent herb
Amaranthaceae						
Sarcocornia cf. natalensis	p	fw+	Indigenous	Herbaceous	IHP	succulent herb
Araceae						
Zantedeschia aethiopica	p	Ow	Indigenous	Herbaceous	IHP	geophytic herb
Asphodelaceae						
Trachyandra filiformis	p	fw+	Indigenous	Herbaceous	IHP	succulent herb
Asteraceae						
Cotula turbinate	a	Fw	Indigenous	Herbaceous	IHAn	herb
Senecio littoreus	a	Fw	Indigenous	Herbaceous	IHAn	herb
Caryophyllaceae						
*Spergularia media**	p	Fw	Alien	Herbaceous	AHP	succulent herb
Convolvulaceae						
Falkia repens	p	fw+	Indigenous	Herbaceous	IHP	herb
Cyperaceae						
Bolboschoenus maritimus	p	Ow	Indigenous	Graminoid	IGP	graminoid
Cyperus textilis	p	fw+	Indigenous	Graminoid	IGP	graminoid
Isolepis cernua	a	fw+	Indigenous	Graminoid	IGAn	graminoid
Crassulaceae						
Crassula glomerata	a	Fw	Indigenous	Herbaceous	IHAn	succulent herb
Crassula natans	fa	Ow	Indigenous	Herbaceous	IHAn	succulent herb
Fabaceae						
*Acacia cyclops**	p	F	Alien	tree/shrub	AW	small tree

*Vicia benghalensis**	fa-p	F	Alien	Herbaceous		
Haloragaceae						
Laurembergia repens	p	Ow	Indigenous	Herbaceous	IHP	herb
Juncaceae						
Juncus kraussii	p	O	Indigenous	Graminoid	IGP	graminoid
Juncaginaceae						
Triglochin bulbosa	p	Ow	Indigenous	Herbaceous	IHP	geophytic herb
Poaceae						
Cynodon dactylon	p	F	Indigenous	Graminoid	IGP	graminoid
Eragrostis plana	p	Fw	Indigenous	Graminoid	IGP	graminoid
Pennisetum macrourum	p	Ow	Indigenous	Graminoid	IGP	mega-graminoid
Sporobolus virginicus	p	fw+	Indigenous	Graminoid	IGP	graminoid
*Lolium perenne**	p	F	Alien	Graminoid	AGP	graminoid
Polygonaceae						
Persicaria decipiens	a	Ow	Indigenous	Herbaceous	IHAn	herb
Restionaceae						
Elegia tectorum	p	fw+	Indigenous	Graminoid	IGP	graminoid
Scrophulariaceae						
Veronica anagallis-aquatica	fa-p	fw+	Indigenous	herbaceous	IHP	herb
Typhaceae						
Typha capensis	p	Ow	Indigenous	Graminoid	IGP	mega-graminoid

† a= annual, p = perennial, fa = facultative annual, fa-p = facultative annual to perennial (After Goldblatt & Manning 2000)

***I = indigenous, A (at start of group name) = alien, G = Graminoid, H = herbaceous, W = woody, P = Perennial, An = annual

** A + indicates strong conviction of status

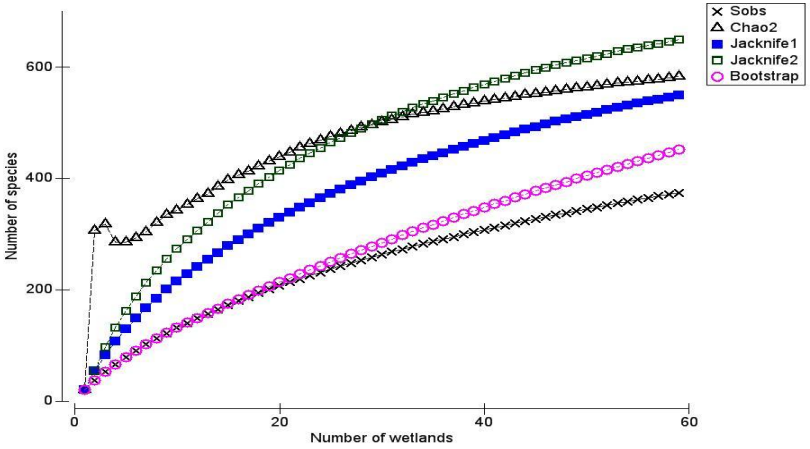


Figure 3.1: Sample-based species accumulation and rarefaction curves of wetland macrophytes for all 59 wetlands sampled on the Cape coastal lowlands. Comparison between the first few samples of a data set with a large number of uniques causes Chao 2 to over-estimate species richness at the beginning of the sample set (Williams *et al.* 2007) (See Section 3.4).

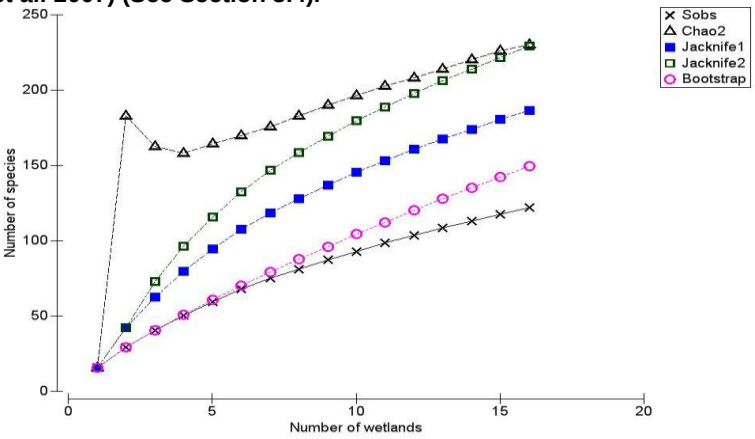


Figure 3.2: Sample-based species accumulation and rarefaction curves of wetland macrophytes for the 16 wetlands sampled on the West Coast, the western sub-region, of the Cape coastal lowlands.

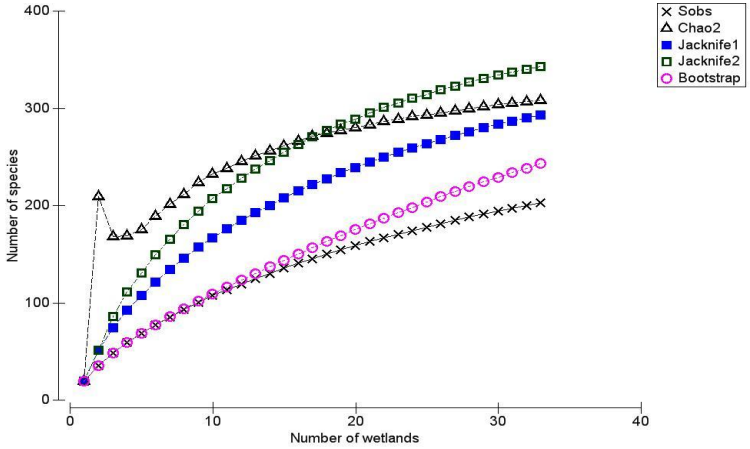


Figure 3.3: Sample-based species accumulation and rarefaction curves of wetland macrophytes for the 33 wetlands sampled on the Cape Flats, in the central sub-region, of the Cape coastal lowlands.

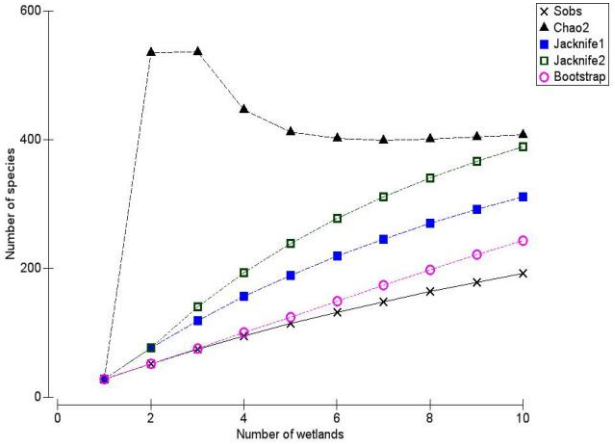


Figure 3.4: Sample-based species accumulation and rarefaction curves of wetland macrophytes for the 10 wetlands sampled on the Overberg, in the eastern sub-region, of the Cape coastal lowlands.

ii. Life-History Group diversity

Comparison of the life-history groups revealed that spatial cover was dominated by indigenous graminoid perennials (IGP) in all of the sub-regions of the Cape coastal lowlands (Figure 3.5). The groups used were similar to those developed by Galatowitsch *et al*. (2000) as based on several major features of the structural and functional life-history of plants such as annual or perennial persistence, origin, and on growth form as described in Section 1.5.5 of the present study.

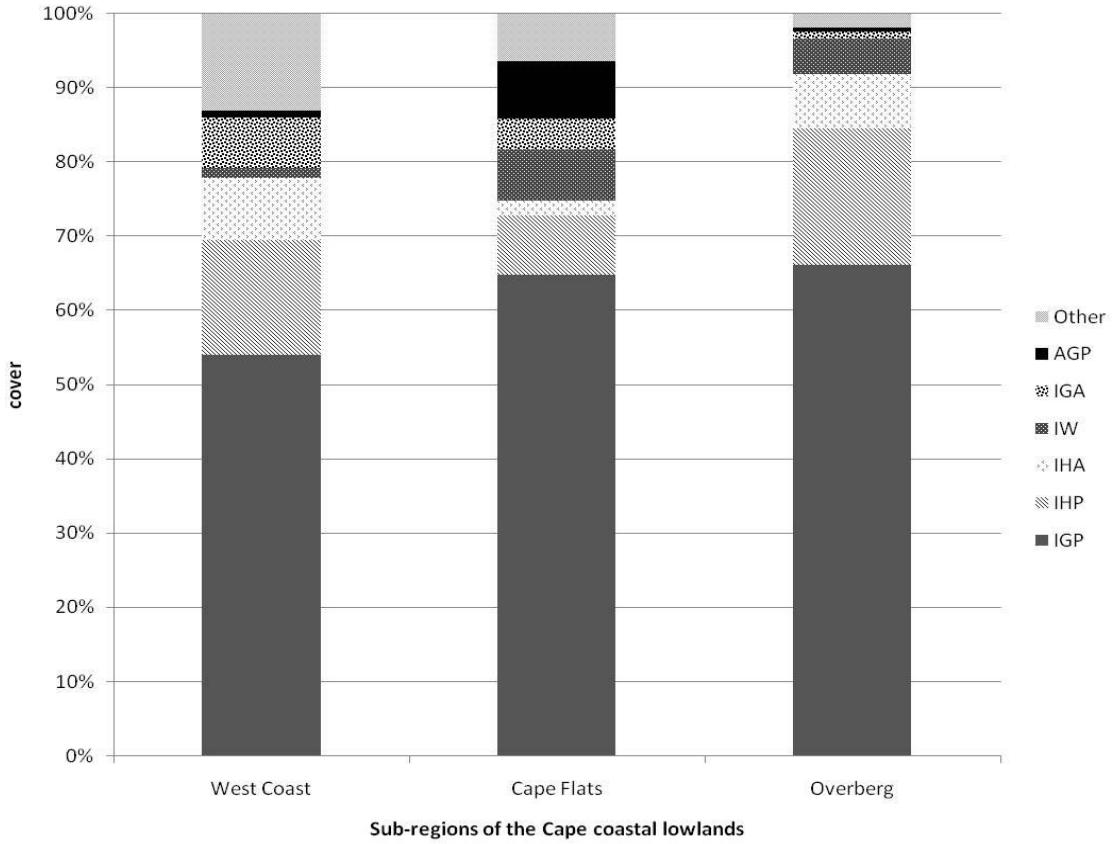


Figure 3.5: Percentage cover of the six dominant life-history groups (*sensu* Galatowitsch *et al.* 2000) and a compilation of seven 'other' less dominant groups in the wetlands of the different sub-regions in the Cape coastal lowlands. IGP: indigenous graminoid perennial, IHP: indigenous herbaceous perennial, IHA: indigenous herbaceous annual, IW: indigenous woody, IGA: indigenous graminoid annual, AGP: alien graminoid perennial, other: combines seven groups with minimal cover, namely and in decreasing order of area covered: alien herbaceous annuals, alien graminoid annuals, macroalgae, alien woody taxa, alien herbaceous perennials, standing dead litter and moss.

Examination of the relative dominance of these life-history groups in reference relative to impaired wetlands across all wetlands surveyed in the Cape coastal lowlands, as ascertained from the similarity of percentage cover, revealed more algae, annuals and alien graminoid perennial taxa in impaired than reference wetlands. More indigenous

woody, indigenous graminoid perennials and indigenous herbaceous perennial taxa were evident in the reference than the impaired wetlands. A graph of the $\log_e$ of the mean percentage cover of those life-history groups that best discriminate between reference and impaired relevés is provided in Figure 3.6.

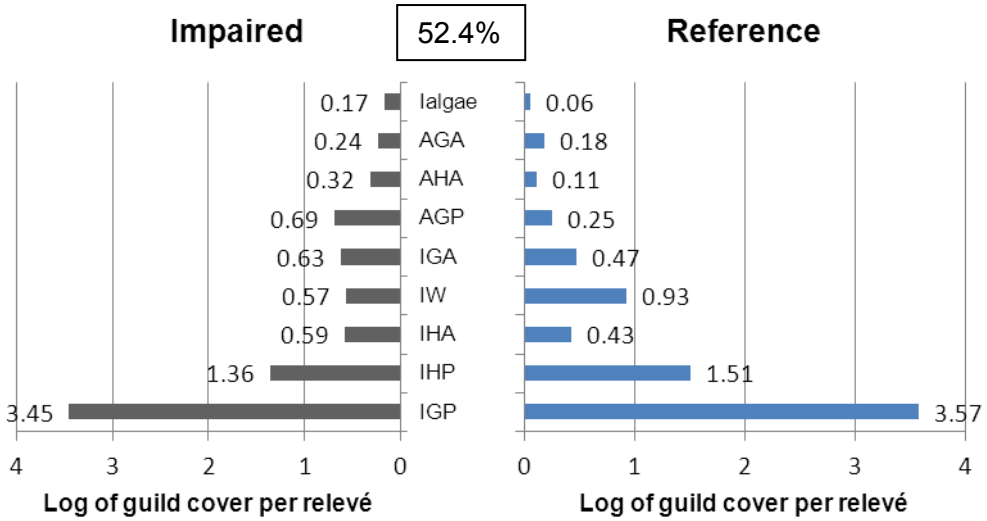


Figure 3.6: A comparison of the $\log_e$ of the average cover of life-history groups (guilds) per relevé in all reference and impaired relevés from the wetlands sampled in the Cape coastal lowlands. The acronyms used are those described in Figure 3.5, other than AHA: alien herbaceous annuals, AGA: alien graminoid annuals and Ialgae: indigenous macroalgae. The percentage of dissimilarity between the guild assemblages of the reference and impaired relevés is presented in the rectangle at the top of the graph.

3.3.2 Hydrological zones

The vegetated aquatic zone was present in only eleven wetlands that were deep enough or inundated for long enough to constitute aquatic habitat. Using this criterion, only 7%, or 27 of the 396 vegetation plots were categorized as aquatic. The aquatic habitat was therefore not sufficiently sampled to facilitate meaningful examination of differences between locations or disturbance groups. The aquatic-zone and littoral zone samples were therefore analyzed together, following the recommendations of the US EPA (2002b), and are referred to throughout as the littoral zone. The supra-littoral zone accounted for 55% (217 of 396) of sample plots and the combination of aquatic (27) and littoral (152) zones accounted for the other 45% of sample plots.

Analysis of similarity between hydrological zones was used to test for differences in terms of floristic community structure in the supralittoral and littoral hydrological zones of the whole data set. A significant but small difference in floristic community structure (small

global *R* value) was apparent between supralittoral and littoral vegetation of the Cape coastal lowlands when wetlands of all sub-regions are collectively examined (R=0.076, p<0.001).

3.3.3 Phytogeographical patterns

Unconstrained ordinations using non-metric Multi-Dimensional Scaling (nMDS) of floristic community structure were used to examine which wetlands or samples are similar as shown by their grouping together in unconstrained space. Testing of the similarity of *a priori* groups of wetlands and or samples was performed with constrained ordination and discriminant analysis as well as analysis of similarity (ANOSIM). Constrained ordinations were performed using canonical analysis of principal coordinates (CAP) to discriminate between a priori determined groups when this was not apparent with nMDS. Considerable differences in species composition (floristic community structure) exist between all wetlands (or samples) in the data set as was evident from the very low Bray-Curtis measure of diversity (0.09; representing a value close to 0 of no similarity) in Table 2.1. The Bray-Curtis measure of dissimilarity is derived from the total number of species that are unique to any one of a number of sites (or in this case, of sub-regions) divided by the total number of species over all sites (or sub-regions) (Bray & Curtis 1957). In other words (and after Legendre & Legendre 2003), it is the ratio between the turnover of species between the sub-regions and the total species richness over all sub-regions.

The low level of similarity between wetlands meant that, using cluster analysis, it was difficult to pick up patterns of compositional similarity relating to geographical proximity of wetlands. Hierarchical agglomerative clustering joining samples based on group average similarity was able to group wetlands with floristic similarity but in this study, in which phytogeographical differences are the focus, broad geospatial patterns were of interest rather than phytosociological similarity and ordination provided a more optimal means of observing such spatial patterns. Clustering analysis was therefore not used, in the present study, beyond initial data exploration efforts.

i. Sub-regions

Unconstrained ordination (nMDS) of the Bray-Curtis dissimilarity (resemblance of community structure) of each wetland suggests difference between the West Coast and Cape Flats wetlands on the first two axes of the ordination as displayed in Figure 3.7, and that the Overberg wetlands are aligned to the third axis or are orthogonal to the first two

axes (i.e. superimposed upon them). The degree to which the species composition of wetlands of the different sub-regions is different is reflected in a constrained ordination (CAP) which shows clear separation of the wetlands from each of the sub-regions of the Cape coastal lowlands, as displayed in Figure 3.8. Separation is apparent in Figure 3.8 between the wetlands of each sub-region, other than a few outliers particularly from the West Coast and Overberg. Discriminant analysis shows that the wetlands of each sub-region are significantly different from those of other sub-regions tr(Q_m'HQ_m): 1.3, p= 0.001. Fifty two of the 59 wetlands (88%) were correctly classified into sub-regions by discriminant analysis of their full species complement.

Figure 3.9 shows an unconstrained nMDS ordination of the similarity of wetlands as based upon the community structure of life-history groups rather than of species. At this broadly incorporative guild-scale in which many species are lumped together, there is more similarity between the wetlands of each area of the Cape coastal lowlands than is apparent at the species scale of examination. The ordination in Figure 3.9 does suggest less difference between West Coast and Cape Flats wetlands than is apparent in Figure 3.8 as separation is no longer apparent between the West Coast and Cape Flats wetlands. Significant differences are however apparent in the life-history resemblance between sub-regions as is apparent with analysis of similarity (See Table 3.4).

Figure 3.7

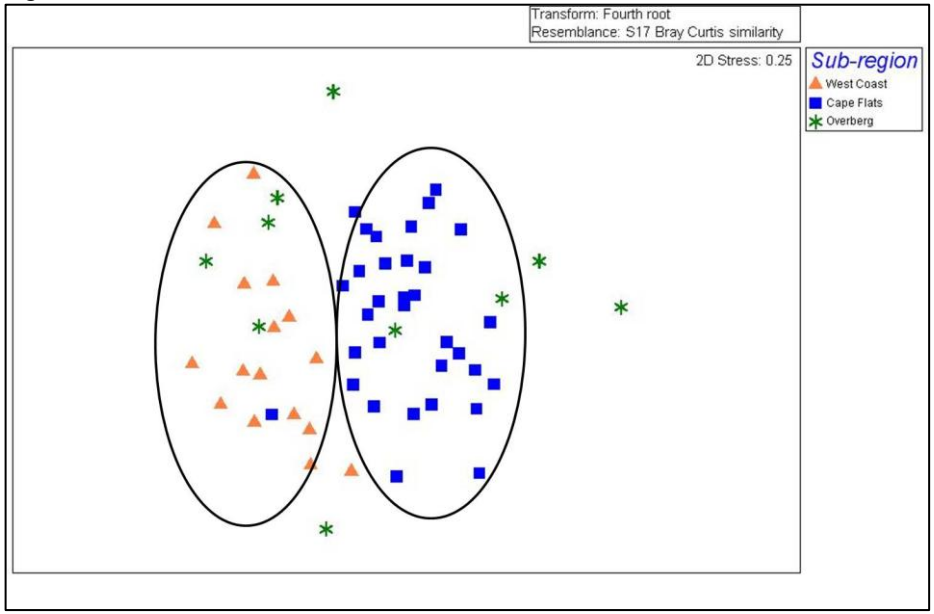


Figure 3.8

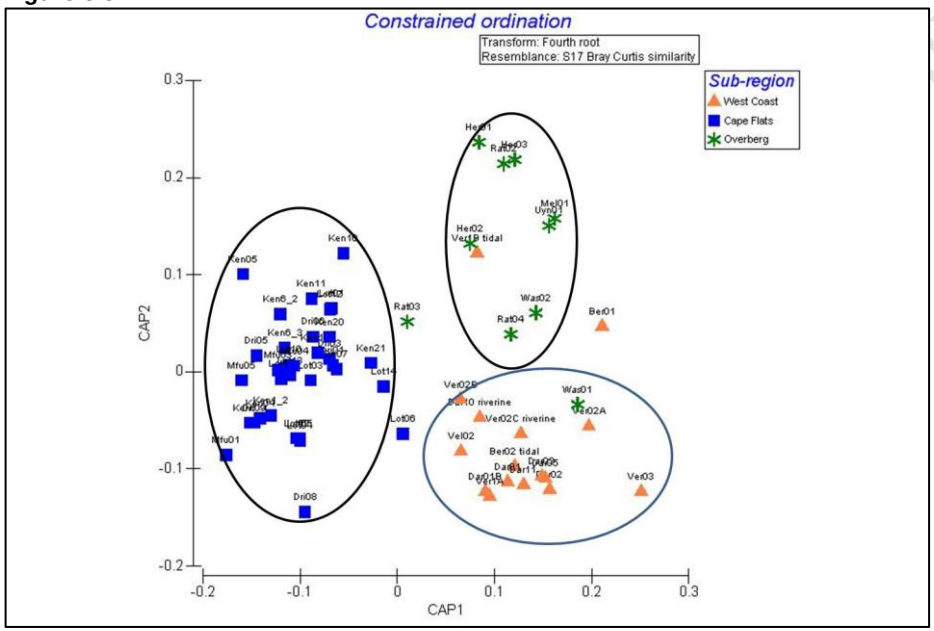
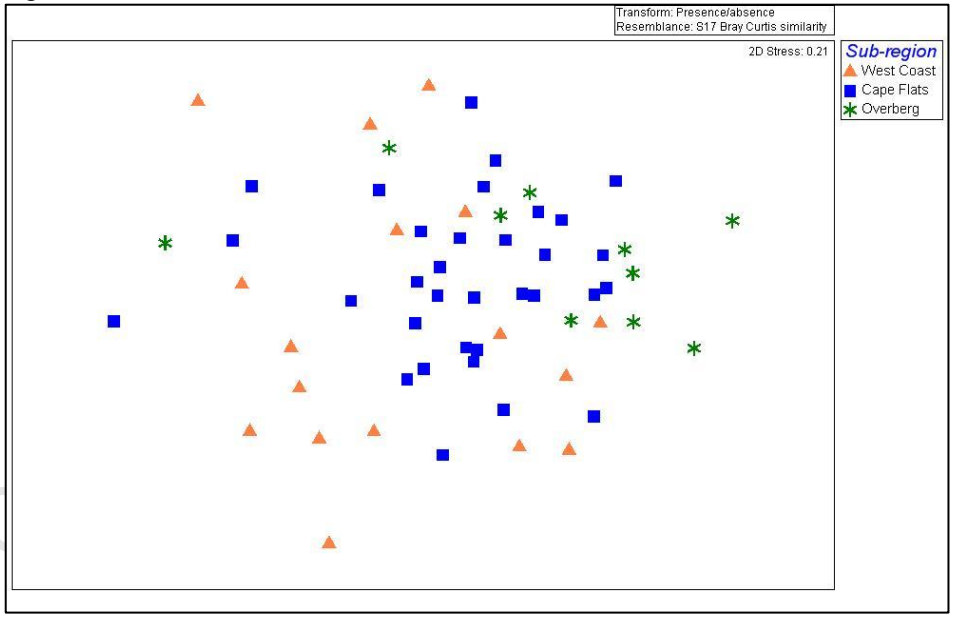


Figure 3.9



Figures3.7: Unconstrained nonmetric Multi-Dimensional Scaling ordination of the plant communities of Cape coastal lowlands wetlands as represented by the Bray-Curtis resemblances of the median cover values per species for the full species complement per wetland.

Figure 3.8: Constrained ordination of the plant communities of 59 wetlands sampled in the Cape coastal lowlands as based on discriminant analysis of the Bray-Curtis resemblance of the full species complement of each wetland.

Figure 3.9: Unconstrained non-metric Multi-Dimensional Scaling ordination of the plant communities of Cape coastal lowlands wetlands as represented by the Bray-Curtis resemblances of the median cover values per life-history group per wetland.

ANOSIM of the differences between sub-regions, based on species and on life-history groups, suggests considerable difference between each of the western (West Coast), central (Cape Flats) and eastern (Overberg) sub-regions (Table 3.4). Each sub-region thus holds significantly different units of wetland vegetation.

Table 3.4: Analysis of similarity of the Bray-Curtis resemblance of the vegetation assemblages for all wetlands. Group comparisons that are significant are marked *.

Pair-wise Tests	R Statistic	*P*-value	Actual Permutations	No. ≥ Observed
Species: Global R = 0.49, p < 0.001*, 999 permutations, # permutations ≥ Global R = 0				
West Coast *vs* Cape Flats	0.482	< 0.001*	999	0
West Coast *vs* Overberg	0.375	< 0.001*	999	0
Cape Flats *vs* Overberg	0.556	< 0.001*	999	0
Life-history groups: Global R = 0.256, p < 0.001*, 999 permutations, # permutations ≥ Global R = 0				
West Coast *vs* Cape Flats	0.294	< 0.001*	999	0
West Coast *vs* Overberg	0.267	< 0.005*	999	4
Cape Flats *vs* Overberg	0.168	< 0.04*	999	40

ii. Bioregions

Unconstrained ordination (nMDS) of the species assemblages in wetlands in each of the bioregions (zones of similar climate) shows considerable distinction between the wetlands of the South-West Fynbos (in the dashed ellipse) and West Strandveld (in the solid ellipse) as displayed in Figure 3.10, suggesting that the climatic and geological differences in each bioregion are potentially driving differences in species distribution between bioregions. A constrained ordination (CAP) of the same data suggests significant difference between the wetlands of all bioregions as is apparent due to the separation between the wetlands of each bioregion in Figure 3.11. Discriminant analysis shows that the wetlands of each bioregion are indeed significantly different from those of other bioregions tr(Q_m'HQ_m): 4.3, p= 0.001.

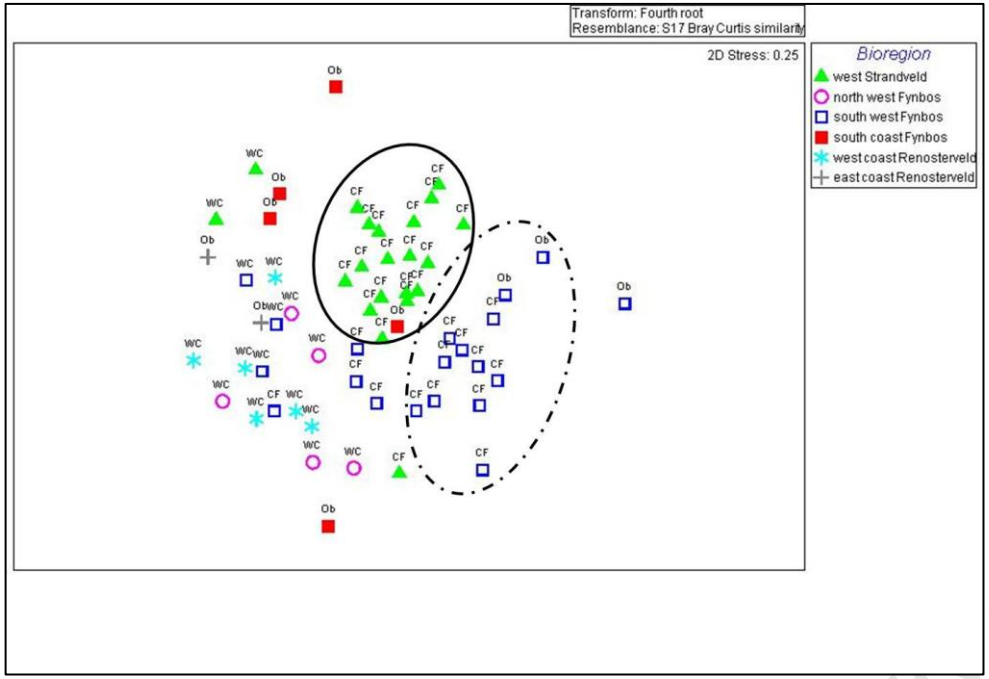


Figure 3.10: Non-metric Multi-Dimensional Scaling ordination of the wetlands of the Cape coastal lowlands as represented by the Bray-Curtis resemblances of their full species complement. Considerable difference is apparent between the wetlands from the South-west Fynbos and West Strandveld bioregions.

Based on their community structure (species complement) 45 of the 59 (76%) wetlands were correctly classified into bioregions by the discriminant analysis performed to produce the constrained ordination in Figure 3.11.

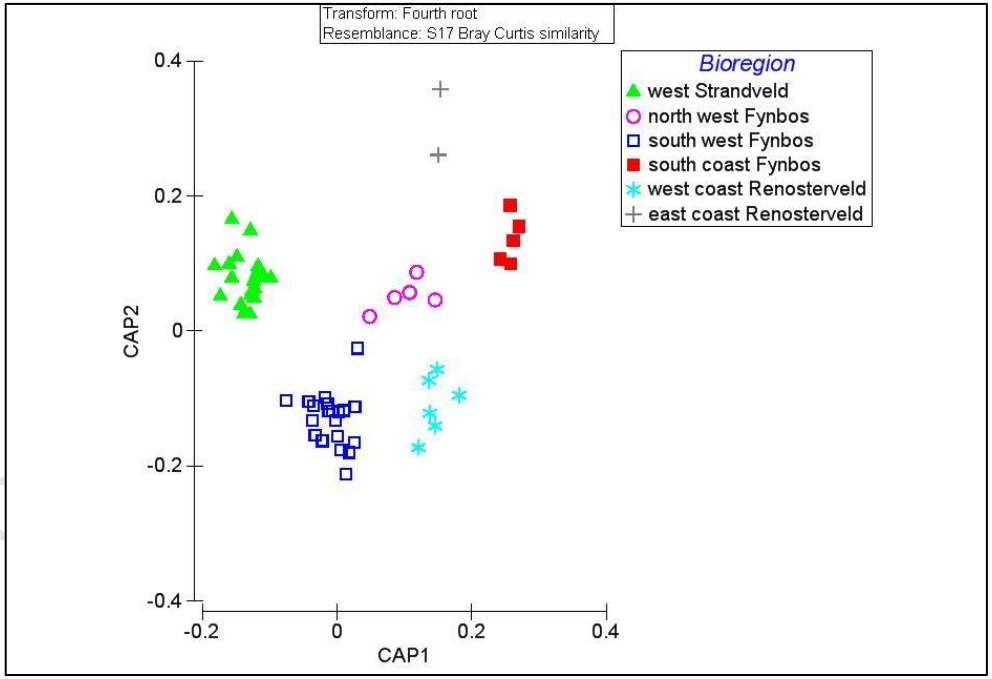


Figure 3.11: Constrained ordination (CAP) of the 59 wetlands sampled in the Cape coastal lowlands as based on discriminant analysis of the Bray-Curtis resemblances of the community structure of each wetland.

Examination of the floristic differences between the wetlands associated with each of the terrestrial vegetation units incorporated into the South-West Fynbos bioregion showed considerable spatial disjunctions suggestive of significant differences between each unit when examined with unconstrained ordination (nMDS) as is evident in Figure 3.12.

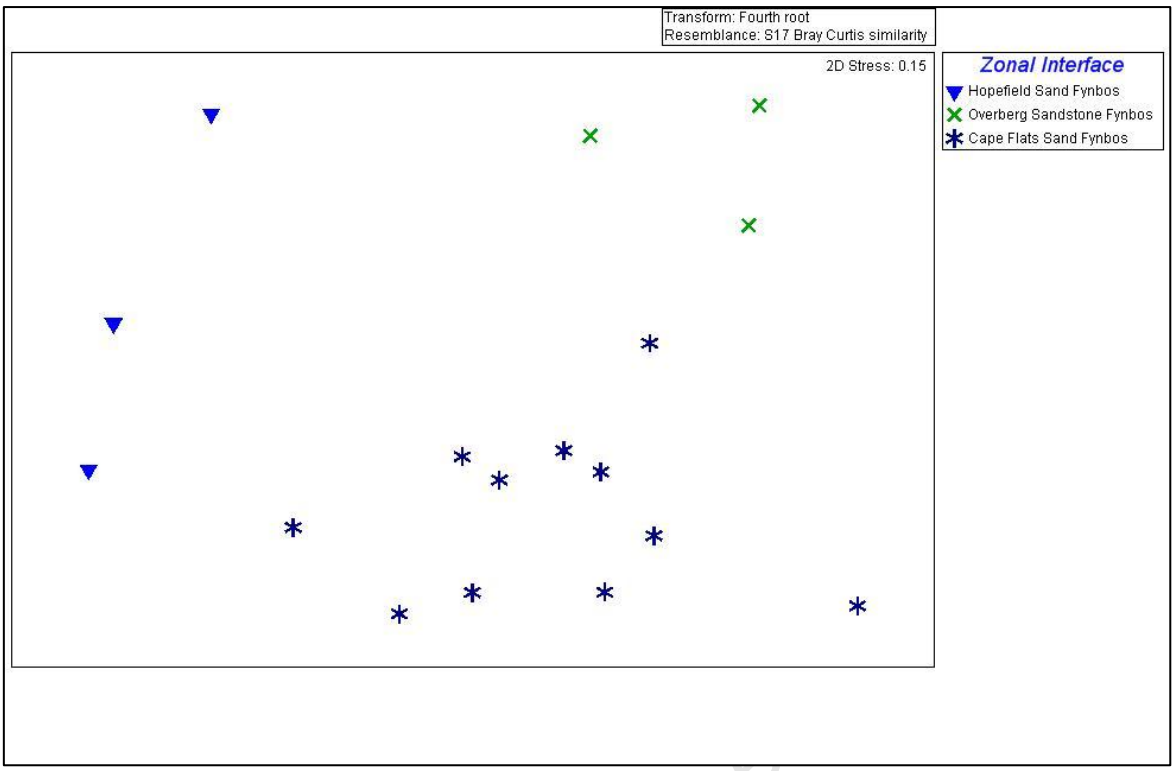


Figure 3.12: Non-metric Multi-Dimensional Scaling ordination of the wetlands of the South-West Fynbos Bioregion as represented by the Bray-Curtis resemblances of their species complement. Considerable disjunction and thus difference is apparent between wetlands associated with each of the different terrestrial or zonal units of vegetation.

The influence of environmental variables (both climatic and edaphic, as listed in Table 3.5) on floristic community composition of the wetlands from different terrestrial vegetation units within the South-West Fynbos bioregion was assessed using a distance-linear model (DistLM, selection criteria "AICc" and "Best" procedure). None of the environmental variables were collinear at greater than 90% as determined in a search for multi-collinearity; hence all variables were retained in the DistLM (Anderson *et al.* 2008).

Table 3.5: Summary of average edaphic and climatic variables (±Standard Error) for each of the wetland vegetation units associated with a different terrestrial unit of vegetation within the South West Sand Fynbos bioregion.

Sub-region:		West Coast	Cape Flats	Overberg
Associated terrestrial vegetation unit:		Hopefield Sand Fynbos	Cape Flats Sand Fynbos	Overberg Sandstone Fynbos
Variables	Units			
% Clay in soil	%	5.8 ± 4.8	1.4 ± 0.1	6.5 ± 0.3
% Nitrogen in soil	%	0.1 ± 0.1	0.1 ± 0	0.2 ± 0
% Carbon in soil	%	0.8 ± 0.7	1.1 ± 0.1	6.1 ± 1
Bulk Density	$kg.m^{-3}$	1.5 ± 0.1	1.5 ± 0	1.5 ± 0
pH of soil	pH units	7.8 ± 0.5	5.0 ± 0.2	3.7 ± 0.4
soil resistance	Ohms	398 ± 225	2402 ± 341	367 ± 23
[Phosphorus]	$mg.kg^{-1}$	25.28 ± 18.65	65.1 ± 34.2	16.5 ± 2.25
[Potassium]	$mg.kg^{-1}$	311.8 ± 258.6	13.05 ± 1.85	67 ± 10
Cation Exchange Capacity	$cmol_c.kg^{-1}$	4.9 ± 2.9	2.75 ± 0.18	2.75 ± 0
Minimum daily temperature	^{o}Celsius	11 ± 0	11 ± 0	10 ± 0
Max daily temperature	^{o}Celsius	23 ± 0	21 ± 0	22 ± 0
Annual Evaporation	$mm.a^{-1}$	2291 ± 28	1953 ± 13	1922 ± 0
Annual Rainfall	$mm.a^{-1}$	278 ± 26	1040 ± 15	587 ± 0

A number of the variables ([K], [P], & resistance) had skew distributions and were therefore log_e transformed before performing the DistLM as recommended by Anderson *et al.* (2008).

The DistLM revealed that all the climatic variables and the carbon content, pH, resistance and potassium concentration in the soil were all significantly correlated with the distribution of flora in the South West Fynbos wetlands (see Marginal Tests in Table 3.6). All other variables resulted in non-significant correlations ($p \geq 0.05$). Rainfall and pH of the soil, and/or rainfall and minimum temperature, and/or rainfall alone each represent variables that correlate most highly and significantly with the distribution of the wetland flora in the South-West Fynbos bioregion as is evident from the overall best solutions from the DistLM in Table 3.6.

A distance-based redundancy analysis (dbRDA, a constrained ordination of the fitted values from the linear model, with an overlay of those variables with strongest correlations to the distribution of the samples) based on the environmental variables selected in the DistLM model indicates that 30% of the total variation in the ordination was explained by rainfall and soil pH. The dbRDA is displayed in Figure 3.13.

Table 3.6: Test statistics for a Distance-based Linear model (DISTLM) analysis based on "Best" procedure and "AICc" criteria of the influence of edaphic and climatic variables on the floristic community structure differences of wetlands in the South-West Fynbos Bioregion. SS= Sum of Squares, RSS = residual Sum of Squares, R^2 = RSS/SS

MARGINAL TESTS:

Variable	SS(trace)	Pseudo-F	p value	% of variation
Annual Rainfall	8997	2.81	0.001	15.78
Annual Evaporation	8097	2.48	0.002	14.20
Max daily temperature	8691	2.70	0.001	15.25
Minimum daily temperature	7922	2.42	0.005	13.90
% carbon in soil	7678	2.34	0.002	13.47
pH of soil	8189	2.52	0.003	14.37
(soil resistance) $\log_e$	7607	2.31	0.003	13.35
[Potassium]$\log_e$	6757	2.02	0.008	11.86
% Clay in soil	5325	1.55	0.06	9.34
Bulk Density	3824	1.08	0.37	6.71
[Phosphorus]$\log_e$	3982	1.13	0.33	6.99
% Nitrogen in soil	4752	1.36	0.12	8.34
Cation Exchange Capacity	3959	1.12	0.39	6.95

OVERALL BEST SOLUTIONS

AICc	R^2	RSS	No. of Var's	Selections
139.78	0.29997	39902	2	rainfall & soil pH
139.92	0.29422	40229	2	rainfall & minimum temperature
139.94	0.15784	48003	1	rainfall

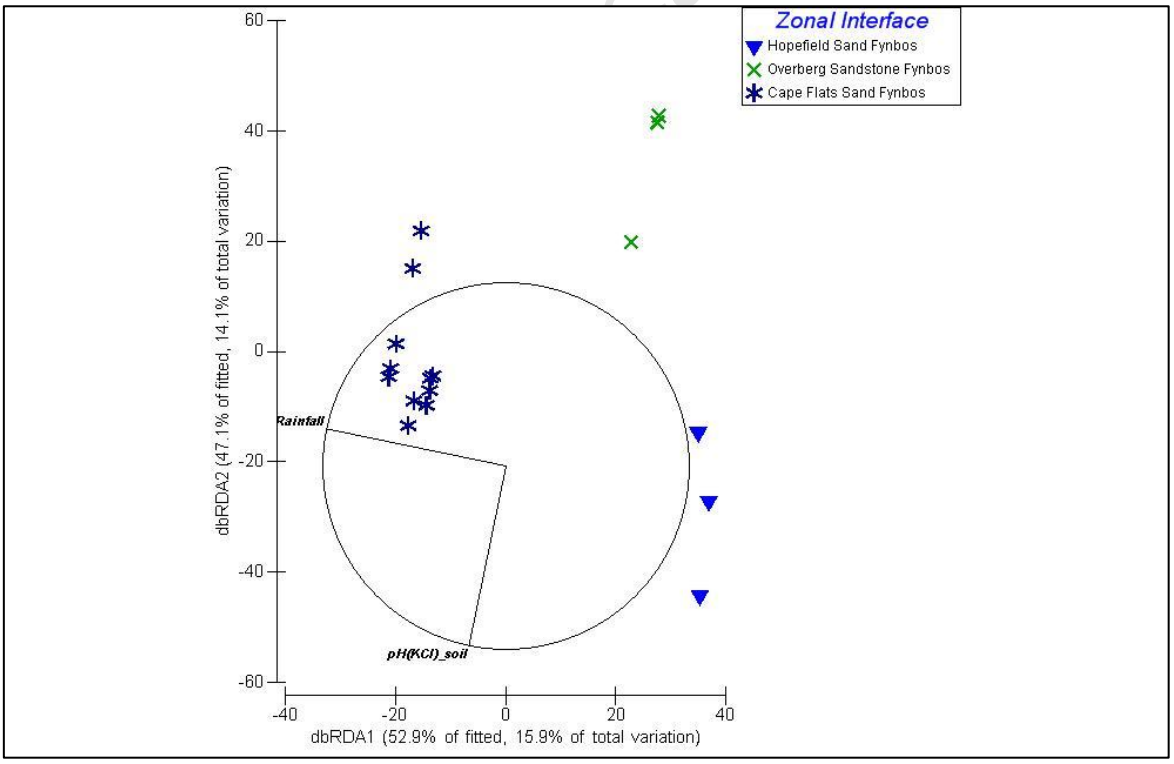


Figure 3.13: Distance-based Redundancy Analysis of the 4th root transformed Bray Curtis similarity of the floristic composition of wetlands associated with different terrestrial vegetation units within the South-West Fynbos Bioregion. The vector overlays are of the rainfall and soil pH that best fit the differences in distribution.

Differences in annual rainfall in the Cape Flats, Overberg and Hopefield (West Coast) areas of the South-West Fynbos bioregion are significant; as is the difference in soil pH values between Hopefield and the Overberg wetlands. It would therefore seem likely that the climatic or moisture gradient certainly effects the distribution of wetland plant species within the South-West Fynbos bioregion and may equally be expected to do so in other bioregions that straddle the same gradient. It is also apparent from these results that edaphic or geological factors are also significantly correlated with this floristic distribution pattern.

iii. National Freshwater Ecosystem Priority Areas

The difference between the National Freshwater Ecosystem Priority Areas (NFEPA) wetland vegetation groupings can also be shown with ordinations or with ANOSIM. Of the 13 terrestrial vegetation units (*sensu* Rebelo *et al*. 2006) within which wetlands were sampled in the present study (see Table 2.2), six were incorporated into two of the NFEPA vegetation groups, namely into the "*Western Strandveld"* and "*South-west Sand Fynbos".* These groups contain Strandveld vegetation units from the Cape Flats, Lamberts Bay and Saldanha Flats and Sand Fynbos vegetation units from the Cape Flats, Atlantis and Hopefield. The remaining seven terrestrial vegetation units in which wetlands were sampled in this study each represent the only unit included within different NFEPA vegetation groups.

Comparison of the wetlands sampled in the present study from *Western Strandveld* and *South-west Sand Fynbos* NFEPA groups is essentially a comparison of the South-West Fynbos and West Strandveld Bioregions (of Rutherford *et al.* 2006) other than the Overberg Sandstone Fynbos which would be excluded as it belongs to the *South-West Sandstone Fynbos* NFEPA group. Comparison of these wetlands performed in the previous section showed significant floristic differences between the Bioregions and thus by correlation equally shows difference between these NFEPA groups.

ANOSIM of the NFEPA vegetation groups the *South-west Sand Fynbos* and the *North-west Sand Fynbos* representing Leipoldtville Sand Fynbos shows significant difference when all of the terrestrial units of the *South-west Sand Fynbos* are included ($R = 0.2$, $p < 0.01$). When the community structure of wetland vegetation from each of the individual terrestrial vegetation units of Sand Fynbos are compared with ANOSIM, greater resemblance is apparent among the Atlantis, Hopefield and Leipoldtville units, all of which are from the West Coast sub-region and have similar climatic constraints, than between

any of these and the Cape Flats unit which represents a wetter climatic sub-region of the Cape coastal lowlands (Table 3.7).

Table 3.7: Analysis of similarity of the wetland vegetation community associated with each terrestrial vegetation unit incorporated into the South West and North West Sand Fynbos vegetation groups of NFEPA. Significant differences are marked with an asterisk:*

Terrestrial vegetation units compared	R-value	p-value
Hopefield Sand Fynbos, Cape Flats Sand Fynbos	0.3	0.001*
Atlantis Sand Fynbos, Cape Flats Sand Fynbos	0.3	0.01*
Leipoldtville Sand Fynbos, Cape Flats Sand Fynbos	0.3	0.001*
Hopefield Sand Fynbos, Atlantis Sand Fynbos	0.2	0.2
Hopefield Sand Fynbos, Leipoldtville Sand Fynbos	0.02	0.3
Atlantis Sand Fynbos, Leipoldtville Sand Fynbos	0.1	0.2

iv. Zonal interface with wetland vegetation

The floristic community structure of wetlands associated with each of the terrestrial or zonal vegetation units assessed in the Cape coastal lowlands are considerably different from each other as is apparent using unconstrained ordination (nMDS) as displayed in Figure 3.14 and more apparent using constrained ordination (CAP) (Figure 3.15 and Figure 3.16). Discriminant analysis shows that the wetlands associated with each terrestrial vegetation unit are significantly different from each other tr(Q_m'HQ_m): 5.1, p= 0.001. Based on their species complements 42 of the 59 wetlands (71%) were correctly classified by their association with a terrestrial vegetation unit. With constrained ordination, the wetland vegetation assemblages associated with the Cape Flats Dune Strandveld are most easily distinguishable from the rest of the wetland vegetation units as is evident in Figure 3.15.

Four wetlands in the Lotus River area of the Cape Flats (those indicated with hollow circles in Figures 3.14 and 3.15) were geographically located at the interface between terrestrial Strandveld and Fynbos units of vegetation. These wetlands have different community structure from either the Cape Flats Dune Strandveld or Sand Fynbos wetlands. Lot14 was a heavily disturbed wetland and its position relative to the other wetlands from the interface between Fynbos and Strandveld, which were less disturbed (lower HDS scores), reflects this difference in both Figures 3.14 and 3.15. Wetland Lot14 was grouped with Fynbos associated wetlands Lot06 and Ken21 in the discriminant analysis performed to create the constrained ordination in Figure 3.15. Lot06 was, like Lot14, a heavily disturbed wetland as ascertained by the degree of human landuse (HDS) within and surrounding these wetlands. Ken21 had a lower HDS score but had considerable physical disturbance due to repeated mowing of the vegetation.

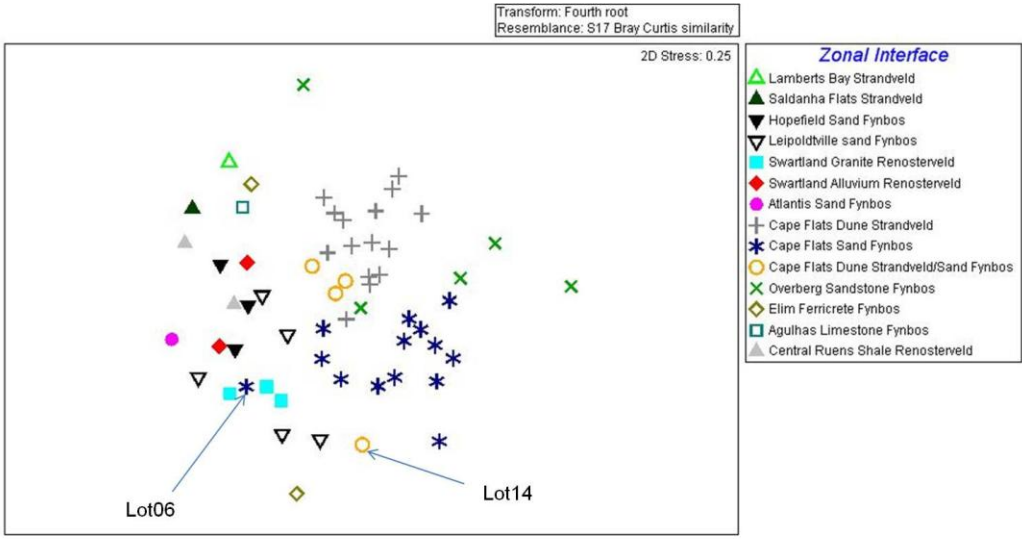


Figure 3.14: Multi-Dimensional Scaling ordination of the wetlands of the Cape coastal lowlands as represented by the Bray-Curtis resemblances of their community structure. Considerable grouping of, and thus difference between, the wetlands associated with many of the different terrestrial vegetation units is apparent. The labelled wetlands are discussed in the text below.

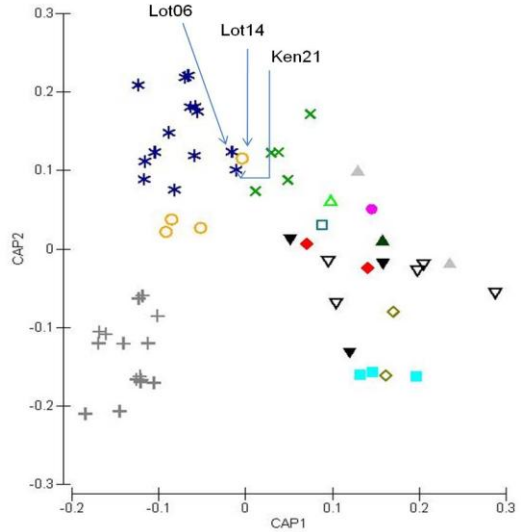


Figure 3.25: Constrained ordination of the same wetlands as in Figure 3.14, as based on discriminant analysis of the Bray-Curtis-resemblances of species complement.

As the wetlands of the Cape Flats Dune Strandveld area were clearly different from those associated with other terrestrial vegetation units in the constrained ordination in Figure 3.15 a second ordination was run after removing these wetlands from the dataset. Based on their community structure 28 of the remaining 44 wetlands (63%) were correctly classified by their association with a terrestrial vegetation unit. Discriminant analysis showed that the wetlands associated with each remaining terrestrial vegetation unit were significantly different from each other tr(Q_m'HQ_m): 6.8, p= 0.001.

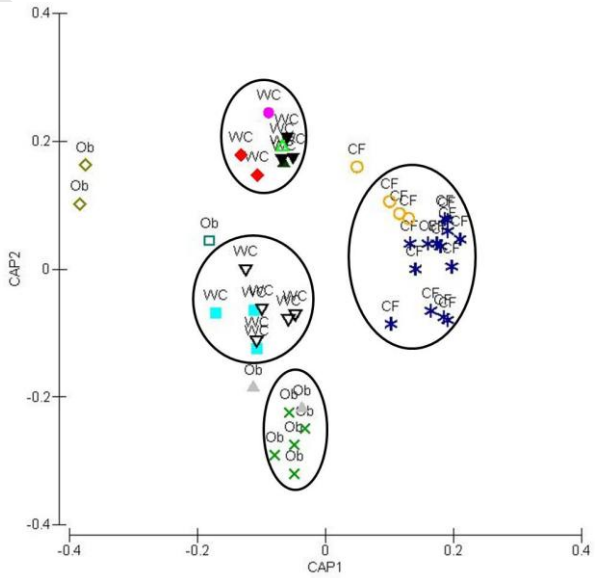


Figure 3.16: Constrained ordination of the forth-root-transformed-Bray-Curtis-resemblances of the full species complements of 44 wetlands sampled in the Cape coastal lowlands after exclusion of the Cape Flats Dune Strandveld wetlands. The symbols correspond to the legend presented in Figure 3.14 and the initials refer to West Coast, Cape Flats or Overberg sub-regions.

Three wetlands from the Overberg sub-region, two associated with Elim Ferricrete Fynbos and one with Agulhas Limestone Fynbos, were outliers from their sub-region in Figure 3.16 by virtue of the fact that their

vegetation differed considerably from that of any other Overberg wetland. These three wetlands were the only limestone and ferricrete associated wetlands in the study. The other two wetlands outside of the ellipses are not considered as outliers.

3.3.4 Biotic exchange and homogenization

A decreasing number of indigenous species and an increasing number of alien species were found to correspond with increasing levels of disturbance as measured by the Human Disturbance Score (HDS). These trends are displayed in Figure 3.17. It is important to keep in mind that the extent of alien vegetation cover within wetlands was not used in the development of the Human Disturbance Scores (HDS). The number of indigenous species showed a negative correlation to increasing disturbance (Pearson's correlation coefficient r = -0.25, being a medium effect size (Cohen 1988)) and the number of alien taxa showed a positive correlation (Pearson's r = 0.11, being a small effect size). The ratio of the number of alien to indigenous taxa shows a far stronger correlation to HDS (r = 0.42) and the graph of this in Figure 3.18 shows an increasing number of alien species relative to the number of indigenous species with increasing human disturbance. An increasing percentage cover of alien relative to indigenous taxa positively correlates with increasing human disturbance (Pearson's r = 0.42) as displayed in Figure 3.19.

In Figures 3.17 to 3.19 a related effect size to Pearson's r is displayed, namely the coefficient of determination that represents the square of r, or "r-squared". The r^2 is positive, so does not convey the polarity of the relationship between the two variables. For the HDS and species variables, this is a measure of the proportion of variance shared by the two variables, and varies from 0 to 1. The relationship of the number of alien and/or indigenous species to disturbance is weak as shown by the low coefficients of determination (r^2 values) for these variables (Figure 3.17). The number of alien species increases relative to the number of indigenous species with increasing disturbance and nearly 18% of the variance of HDS is shared with the variance in the ratio of the number of indigenous to alien species (r^2 = 0.18; Figure 3.18). The large coefficient of determination suggests that this relationship is strong. A similarly strong relationship was shown by the ratio of increasing cover of alien relative to indigenous species with increasing disturbance (r^2 = 0.177; Figure 3.19).

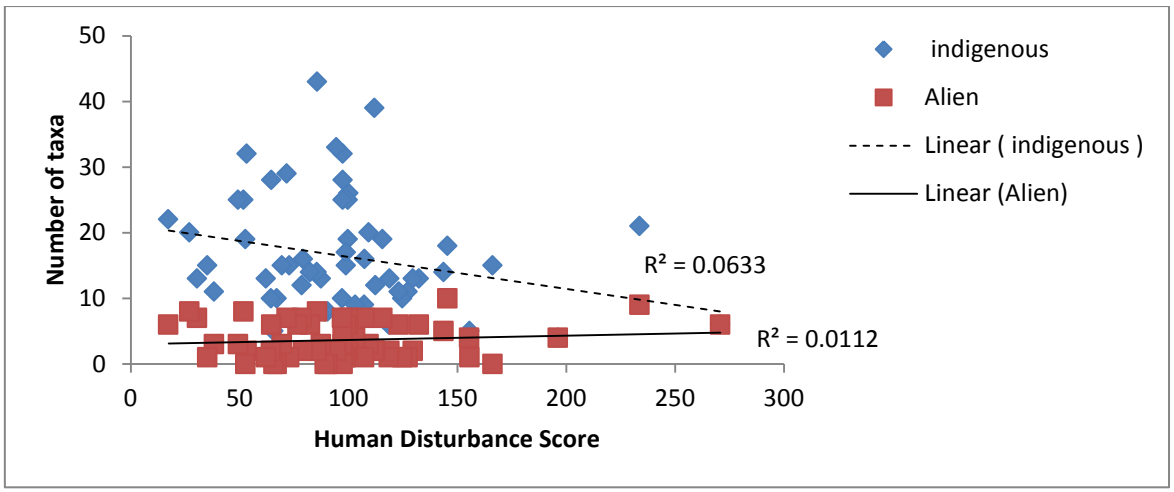


Figure 3.17: The number of alien and indigenous taxa in wetlands with different amounts of human disturbance.

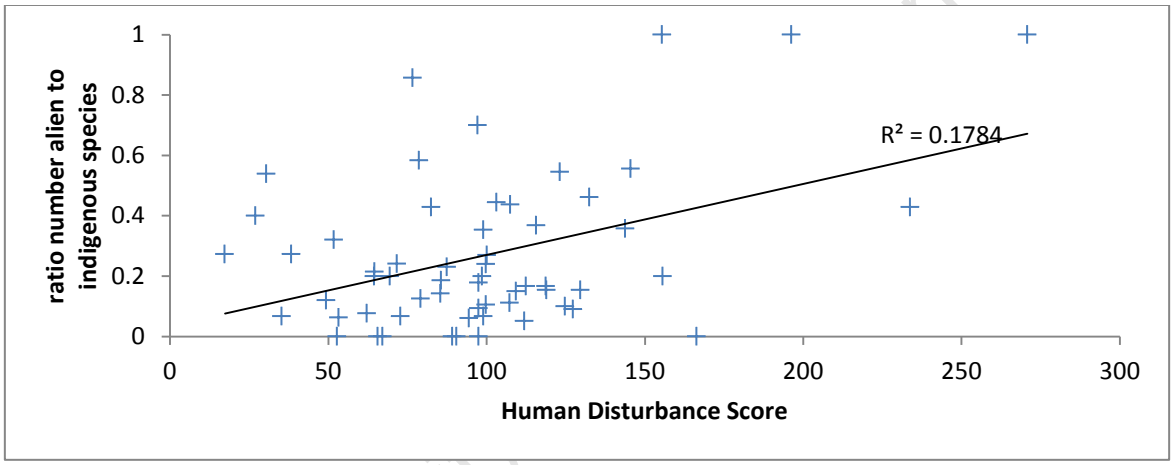


Figure 3.18: The ratio of the NUMBER of alien to indigenous taxa in wetlands with different levels of human disturbance. The number of alien species increases relative to the number of indigenous species with increasing disturbance. A strong relationship is indicated (r^2 = 0.18).

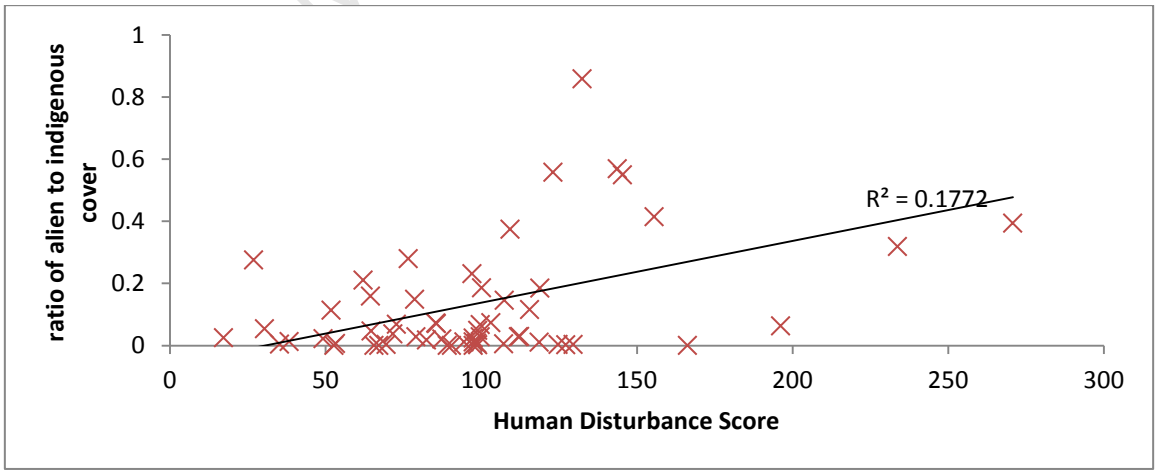


Figure 3.19: The ratio of the COVER of alien to indigenous taxa in wetlands with different levels of human disturbance (17.72% of the co-variation is explained by the coefficient of determination).

The flora of minimally impaired (reference) and impaired wetlands is considerably different in the Cape Flats data set as determined using analysis of variance ($F_{1, 31}$ 17.1, p = 0.003). Ordination of the Bray-Curtis resemblances of these wetlands reveals greater spread or dispersion of the impaired than of the reference wetlands (associated with each of the vegetation units) as shown in Figure 3.20. Examination of the amount of floristic variation (as measured by dispersion from centroids i.e. the amount of spread in the ordination) of the impaired and reference wetlands on the Cape Flats shows that reference wetlands have less variation (wetlands within central ellipses have greater similarity to each other) than the impaired wetlands (within the outer ellipses). The same pattern is apparent for reference and impaired wetlands at the interface of the Strandveld and Fynbos terrestrial vegetation units as indicated with open and closed dots in Figure 3.20.

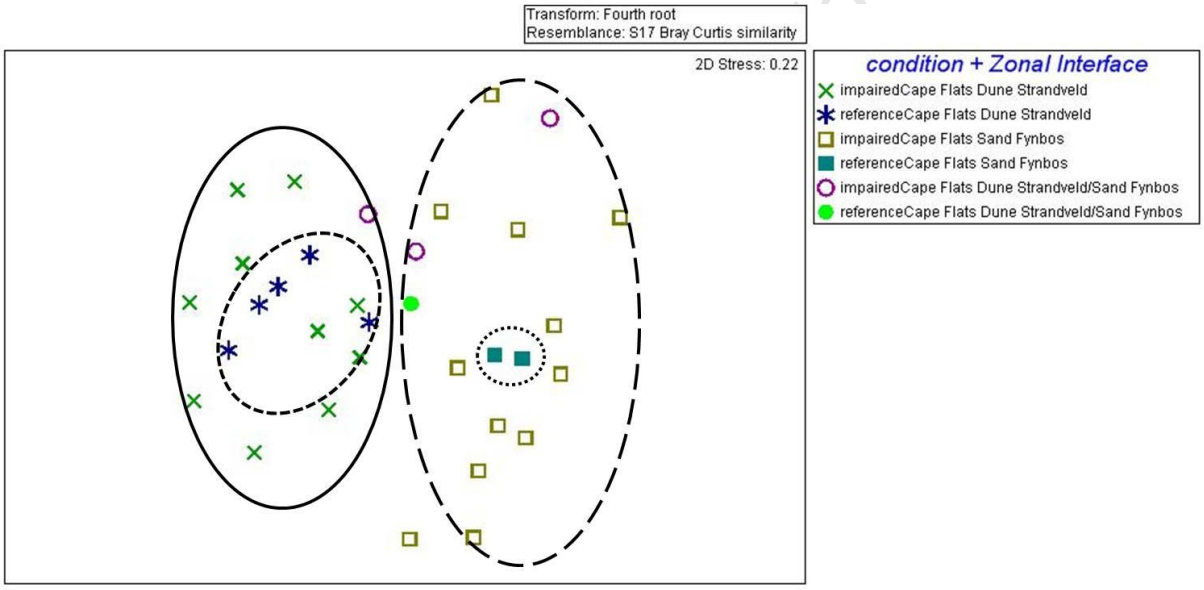


Figure 3.20: Multi-Dimensional Scaling ordination of the wetlands of the Cape Flats as represented by the Bray-Curtis resemblances of their floristic community structure. Wetlands associated with each of the terrestrial (zonal) vegetation units (including wetlands at the geographical interface between two of the zonal units) are displayed with a different symbol. Considerable difference is apparent in the spread (dispersion) of the reference wetlands, within the small central ellipses, relative to the impaired wetlands, within the large outer ellipses.

Dispersion therefore can be said to increase with disturbance, thereby reducing overall floristic differences between impaired Fynbos and Strandveld associated wetlands. This difference of internal variation was tested using individual student t-tests with Monte Carlo permutational determination of significance (Hope 1968) as displayed in Table 3.8.

Table 3.8. Difference of internal variation of the floristic resemblance within groups of reference and impaired wetlands associated with the Cape Flats Dune Strandveld and Sand Fynbos terrestrial vegetation units. Significant comparisons are marked with an asterisk (*)

Groups	t	p(perm)
Impaired Cape Flats Dune Strandveld, reference Cape Flats Dune Strandveld	2.8	0.006*
Impaired Cape Flats Sand Fynbos, reference Cape Flats Sand Fynbos	4.5	0.015*
Impaired Cape Flats Dune Strandveld, impaired Cape Flats Sand Fynbos ††	1.5	0.18
Reference Cape Flats Dune Strandveld, reference Cape Flats Sand Fynbos ††	2.4	0.19

†† No difference in amount of variation (dispersion from centroids) among the impaired and among the reference wetlands

The wetlands associated with both of the Cape Flats Dune Strandveld and Sand Fynbos vegetation units have significantly different floristic variation within reference than within impaired wetlands. When examined independently the impaired wetlands and the reference wetlands from Dune Strandveld and / or from Sand Fynbos each have similar levels of floristic variation (dispersion from centroids). These dispersion differences are representative of beta diversity variation between sets of wetlands (Anderson *et al.* 2008). In other words significant variation in the identities of species exists between reference and impaired wetlands of the same vegetation type.

Analysis of similarity of the impaired wetlands of these Strandveld and Fynbos units and of their reference wetlands would reveal a greater difference (global *R* value closer to 1) between reference than between impaired wetlands if homogenization was occurring as a result of disturbance. Analysis of similarity of the impaired wetlands suggests a large and considerable difference between Strandveld and Fynbos wetlands with an *R* value of 0.6 and a significance level $p<0.001$. The difference between the reference wetlands of Dune Strandveld and Fynbos was, as expected, even greater ($R=0.9$, $p<0.05$). Floristic difference between reference Strandveld vs impaired Fynbos associated wetlands was lower ($R = 0.56$, $p <0.01$) than that between reference Fynbos and impaired Strandveld ($R = 0.76$, $p <0.02$).

In general, on the Cape Flats, human disturbance is thus reducing the difference that naturally existed between minimally impaired (reference) wetlands associated with Strandveld and Fynbos.

3.3.5 Supralittoral vegetation of HGMs

The vegetation of wetlands with different HGM types, namely depressions, backwater depressions, valley bottoms and flats was assessed in the present study. For a list of the

HGM types and all classifiable abiotic attributes associated with each wetland see Appendix 2. Only a single HGM type was sampled within any area of land in which edaphic and climatic parameters were homogenous and representative of a homogenous phytogeographical area, namely in each of the sets of wetlands associated with a given terrestrial vegetation unit. It was thus not possible to test whether the supralittoral vegetation of wetlands from different HGMs but within a homogenous phytogeographical region have similar floristic community structure.

3.4 Discussion

Considerable gamma-diversity was apparent in the macrophytes encountered in wetlands of the Cape coastal lowlands. A large number of indigenous and many invasive alien species were encountered with the Cyperaceae, Asteraceae and Poaceae being dominant. Indigenous graminoid perennials dominated the spatial cover in all wetlands. Slightly different flora was apparent between the supralittoral and littoral vegetation zones that represent drier and wetter habitat respectively. Wetlands of the Cape Flats sub-region were sufficiently sampled but the West Coast and Overberg subregions were undersampled. Beta diversity or turnover of species between sub-regions of the Cape coastal lowlands was extensive and suggestive of narrow distribution of many wetland macrophytes. Considerable difference exists between the wetland vegetation of each of the Cape Flats, West Coast and Overberg sub-regions. At a finer spatial scale, significant differences were recorded in the wetland flora of each bioregion (*sensu* Rutherford *et al.* 2006). Significant differences in vegetation were also apparent between the wetlands associated with each terrestrial vegetation unit. The National Freshwater Ecosystem Priority Areas (NFEPA) wetland vegetation groups each incorporate numerous terrestrial vegetation units and thereby incorporate considerable internal floristic variability.

Within the South-West Fynbos, the most comprehensively sampled bioregion, the climatic variables of air temperature and evapotranspiration and soil variables pH, carbon content, resistance and potassium concentration were significantly correlated with the community structure of wetland flora. It is thus apparent that the Cape coastal lowlands are not a homogenous phytogeographical region for wetlands. It is also thus apparent that wetland macrophytes are not azonal and their distribution appears to be associated with similar environmental drivers that determine the distribution of terrestrial plants.

Differences were apparent between the flora of reference and impaired wetlands. Across all wetlands sampled on the Cape coastal lowlands more algae, annuals and alien graminoid perennials were recorded in impaired than in minimally impaired wetlands. More indigenous woody species, indigenous graminoid perennials and indigenous herbaceous perennials were evident in minimally impaired than in impaired wetlands. An increasing percentage cover of alien relative to indigenous taxa positively correlated with increasing human disturbance in the wetlands of this study. At a finer spatial scale, looking at the best sampled sub-region, it is apparent that disturbance has homogenized the vegetation of wetlands of the Cape Flats reducing the differences between Fynbos and Strandveld associated wetlands.

For development of phytoassessment the considerable beta diversity and apparently zonal distribution of wetland vegetation means that species specific indicators developed for one area of the Cape coastal lowlands may not represent the same conditions in other areas or may not even occur in other areas. As a microcosm of the wetland plant diversity in South Africa this suggests that considerable baseline information about the distribution of wetland macrophytes is required before species specific phytoassessment tools can be developed with any sense of confidence that they will be accurate representations of environmental conditions.

3.4.1 Biotic Diversity

i. Species Diversity

Observed taxon richness is unlikely to be a true representation of the species richness within a habitat unless repeated surveys are performed to encompass the full range of seasons when plants are likely to be apparent and identifiable. Sampling bias, such as once-off surveys used in this study, and targeted sampling (based on access to sites) is likely to contribute to the under-representivity of taxa (Magurran 1992, Kent & Coker 1992). The observed species richness (S_{obs}) in this study is thus expected to be an underestimation of the macrophyte richness of the wetlands of the Cape coastal lowlands and its western, central and eastern sub-regions. The observed species richness, the large number of species occurring in single wetlands (uniques) and the low number of observed shared species suggests a considerable richness and diversity of macrophytes within wetlands of the Cape coastal lowlands.

The sample size of 59 wetlands for the Cape coastal lowlands was estimated by the parametric Michaelis-Menten index to be sufficient to capture more than 50% of the estimated species richness. The same index suggests that the wetlands of the Cape Flats were sufficiently sampled, that the West Coast wetlands were less sufficiently sampled and that the Overberg was considerably under-sampled. According to Colwell & Coddington (1994) the low number of shared species in an area, such as the Overberg, and the fact that this area was undersampled, suggests that the richness estimators are likely to show overestimates for species richness and turnover or complementarity (the measure of biotic distinctness). The adjusted Jaccard and Sorenson similarity indices are specifically designed such that undersampling should not affect the outcome of the similarity index (Chao *et al.* 2005) and both suggested that considerable beta diversity or variation did exist between wetlands (sites) within the Overberg as shown by low index values. The observed number of species in the Overberg, recorded from a third of the number of wetlands than in the Cape Flats, was still similar to the number observed in the Cape Flats wetlands and was 40% greater than in the West Coast. This difference (between Overberg and West Coast) is reflective of considerable species variation or beta diversity difference between sub-regions and is even more profound if alien species are excluded from the comparison. After exclusion of aliens both Cape Flats and Overberg have 50% more indigenous species than were observed in the wetlands of the West Coast. The conspectus of all flora of the Fynbos Biome (Goldblatt & Manning 2000) reports a similar, but far weaker, phytogeographical trend with approximately 10% fewer species in the North-West than the South-West phytogeographical centres (See Table 3.10); these numbers do, however, include highland (mountain) as well as lowland species. Whilst most of the wetlands of the present study are within Goldblatt & Manning's (2000) south-western phytogeographical centre of the Cape Floral Kingdom, those from the Agulhas Plain in the Overberg and from the Verlorevlei area on the West Coast represent the North-West and Agulhas Plain phytogeographical centres. Wetlands of the Overberg do therefore show considerable species richness and biotic distinctness; as do the wetlands of the Cape Flats.

The fact that all of the richness estimators predicted considerably higher species richness than that observed suggests that collectively these estimates represent an approximation of the real richness. The 373 species recorded within vegetation samples from wetlands of the Cape coastal lowlands thus represents at best 80% (bootstrap index) of the total number of species that exist. Similar percentages were estimated in all of the sub-regions, despite the inadequate sampling in the West Coast and the Overberg sub-

regions. The number of observed species that were unique to each sub-region, 21 on the West Coast, 109 on the Cape Flats and 90 in the Overberg (Table 3.2), suggests considerable association of species for different geographical areas, thereby contradicting the supposed phytogeographical azonality (Walter 1973, Mucina *et al.* 2006a) of wetland species. The number of unique species within each sub-region is suggestive that establishment or survival limit the spatial distribution of wetland species in the Cape coastal lowlands. As considerable opportunities exist for distribution of propagules of many wetland plants due to the movement of waterfowl (Underhill 1995, Amezaga *et al.* 2002, Clausen *et al.* 2002, Lurz *et al.* 2002, Santamaría 2002) it seems probable that climatic and or edaphic factors limit species establishment or survival in the same way as in the zonal or terrestrial vegetation of the Fynbos biome (Rebelo *et al.* 2006).

Studies undertaken in other mediterranean regions of the world show similar gamma diversity of macrophytes in seasonal wetlands although these studies were often the result of multiple sample seasons and in some cases do not include alien taxa (Table 3.9). The inclusion of multiple sample seasons facilitates both the inclusion of species that are apparent in different seasons of a single year as well as the potential that successional development of communities may allow for changes in species that an ecosystem is able to support (Middleton 1999, 2002) thereby potentially further increasing apparent richness within these former studies.

Table 3.9: Numbers of macrophytes in (predominantly) regional based studies of mediterranean seasonal wetlands

- 407 species in freshwater aquatic ecosystems, dominated by seasonal pools, of Tunisia (Ghrabi-Gammar *et al.* 2009);
- 302 species in temporary pools from the Atlantic Coastal Plains to the Eastern High Plateaux of Morocco (Rhazi *et al.* 2009);
- 174 plant species, within visually homogeneous plots, in 25 temporary ponds in the coastal plain of southwest Portugal (Pinto-Cruz *et al.* 2009);
- 102 species in and around four vernal pools in Mondragó, Mallorca, Spain (Pinya & Gil 2009);
- 62 species (35 helophytes and 27 hydrophytes) in a heterogeneous set of 26 temporary ponds in the Duero river basin of the North Iberian Plateau, Spain (del Pozo *et al.* 2009);
- 140 species (including terrestrial and alien species) in 30 seasonal wetlands in the mediterranean region of Chile (Bliss *et al.* 1998), and
- 150 (Zedler 1987) to 200 (Holland 1976) species (including terrestrial and alien species) that are common in vernal pools of southern California.

Considerable variation in wetland size and volume, as well as in total land surface area incorporated within the sampling universe of each study makes it difficult to compare between studies.

The Bray-Curtis, Jaccard's and Sørenson's similarity indices all returned similar results suggestive of considerable floristic complementarity (turnover or variation) between wetlands across the whole of the Cape coastal lowlands. The large number of unique species within each of the western, central and eastern sub-regions also suggests considerable species turnover or beta diversity variation between these areas (*sensu* Whittaker 1972, Cowling *et al.* 1996) and that some spatial disjunctions exist in species distribution (phytogeography differences). Given the high beta diversity known to exist amongst terrestrial vegetation units of the Fynbos biome (Goldblatt & Manning 2000) this beta diversity between wetland sub-regions is perhaps to be expected. Literature suggests that generally less beta diversity is shown by wetland than upland (terrestrial) flora due to typically more widespread distribution of wetland species (Santamaría 2002). With less than 8% of the wetland plant species of the Cape coastal lowlands being shared across the three sub-regions (i.e. the ratio of all shared species relative to all other species) a narrow distribution range and considerable turnover between sub-regions can be said to exist.

ii Life history Diversity

The greater number of indigenous annuals in wetlands on the West Coast (>30% of all observed species) (See Table 3.10) relative to the number in wetlands on the Cape Flats and the Overberg (≥18%) mirrors the trend of greater annual species diversity in terrestrial phytogeographical centres. For the terrestrial phytogeographical centres, the more arid North-West holds more annuals (10% of all taxa) than the South-West or Agulhas Plain (in both of which annuals comprise 7% of all species) (Goldblatt & Manning 2000).

Table 3.10: Species richness and the proportion of annual species for wetland *vs* all recorded species within the floras of the sub-regions of the Cape coastal lowlands in the context of the phytogeographic centres of the Fynbos Biome (After Goldblatt & Manning 2000 with the addition of wetland data from Table 3.2).

Sub-regions	Area (10^3 km^2)	Phytogeog'c. Centres	Area (10^3 km^2)	Total species		Annuals (% of all)	
				All	**Wetland**	**All**	**Wetland**
West Coast	5	North-west	22	4062	121	415 (10)	45 (37)
Cape Flats	0.25	South-west	23	4654	202	312 (7)	48 (24)
Overberg	0.1	Agulhas Pain	3	1374	191	92 (7)	35 (18)

Species area relationships should not be generated from these area values as they represent broad polygons and not the actual areas sampled in all cases

Although the phytogeographical centres used to measure the terrestrial trend do not correspond exactly to the sub-regions compared in the present study, the trend of an increasing number of annuals moving along the increasing aridity gradient into the north-west is likely to be even more exacerbated if the many seasonal wetlands from the desert region of the Western Coastal Slope (SW.w in Figure 1.1) were to be sampled. The larger proportion of terrestrial, alien and annual species in the West Coast wetlands relative to those from the Cape Flats and Overberg are suggestive of greater levels of perturbation (natural or unnatural) impacting on the West Coast. This difference is potentially due to greater natural seasonal hydrological change on the West Coast, due to a strongly winter-restricted rainfall, relative to those of the other two sub-regions. Many of the studies on American wetlands (Adamus & Brandt 1990, Adamus & Gonyaw 2000) and global wetlands (Sala *et al.* 2000) report a high proportion of annuals and aliens as indicators of perturbation. As an indication of the impacts of human induced wetland perturbation a greater percentage cover of indigenous graminoid annuals, alien graminoid annuals, alien herbaceous annuals and indigenous herbaceous annual species were recorded in the impaired than in the minimally impaired wetlands of the Cape coastal lowlands (Figure 3.6).

3.4.2 Hydrological zones

The test-statistic (R-value) of analysis of similarity (ANOSIM) can suggest significant difference whilst representing an inconsequentially small difference between two groups (Clarke 1993). The significant but small difference shown between the supralittoral and littoral vegetation of the whole data set for the Cape coastal lowlands (R = 0.076) is close to zero, thus suggestive of very limited difference on a scale that ranges from 0 representing no difference to 1 representing complete difference. This is possibly a result of the fact that samples were assigned to *a priori* defined hydrological zones due to the targeted characteristic vegetation stand sampling approach, rather than a transect sampling approach designed to determine hydrological zones (*sensu* Ellery *et al.* 2003 and Sieben *et al.* 2004). This result suggests that the hydrological zonation shown to be important in determining distribution of species within wetlands of Kwa-Zulu Natal (Kotze & O'Connor 2000) (i.e. within the local landscape scale) and in development of phytoassessment tools in the United States of America (US EPA 2002b, Galatowitsch & Van der Valk 1996) is not important when considering phytogeographical differences in wetlands of the Cape coastal lowlands at the (Fynbos) biome scale. Comparison among wetlands of the whole of the Cape coastal lowlands should be performed using the data

set of all species within a wetland and that it would be no more accurate to compare data sets of each hydrological zone. The concepts examined using Figures 3.1 to 3.20 do not therefore need to be re-examined using the cover/abundance of those species sampled only in supralittoral or only in littoral hydrological zones. It is, however, hypothesized that at smaller spatial scales, for instance between wetlands within a given unit of terrestrial vegetation, greater differences may be apparent between the species assemblages of each hydrological zone. In such circumstances separate examination of the assemblages of each zone may facilitate the identification of species with characteristically greater association (indicator species) for one zone than the other or for minimally disturbed *vs* disturbed conditions. This hypothesis will be explored in Chapter 5.

3.4.3 Phytogeography

The establishment of regions of wetland vegetation with sufficient species homogeneity to assist with the recognition of differences in species assemblage caused by human influences rather than by biogeographical differences is of primary concern in the development of phytoassessment tools (Section 1.4.4). The considerable beta-diversity and species turnover in wetlands across the Cape coastal lowlands suggests that this area has too much internal variation to be considered a homogenous region of wetland vegetation. The floristic differences between regional subsets of the Cape coastal lowlands were examined as discussed below to determine the most suitable geographical unit for phytoassessment purposes.

i. Sub-regions

The difference between the West Coast and Cape Flats wetlands of the western and central areas of the Cape coastal lowlands (Figure 3.7 and 3.8) suggests considerable species turnover along geographic gradients, i.e., high beta diversity (Whittaker 1972), and a disjunction in the distribution of wetland species between these two areas. Less difference is apparent when species are lumped together into life-history groups (Figure 3.9). Lumping the species into these groups, with the intention of finding a means of unifying disparate communities in order to assist with the development of phytoassessment tools with broad application, certainly reduces the differences between sub-regions that are apparent with ordination. There is, however, considerable difference between sub-regions when compared using either species or life-history groups as apparent from the results of the ANOSIM (Table 3.4).

ii. Bioregions

The discontinuities in constrained and unconstrained ordinations (Figure 3.10 and 3.11) and the significant differences shown, by discriminant analysis, to exist between the vegetation assemblages of each bioregion suggest that the climatic and geological differences used to characterize each bioregion (Rutherford *et al.* 2006) do influence the geographical distribution of wetland species. This suggests that these environmental differences present a geographical gradient causing beta diversity differences or turnover in floristic community structure of wetlands between the bioregions. The data therefore suggests that the geographical distribution of wetland vegetation in the Cape coastal lowlands is thus likely to be affected by macroclimatic factors and cannot be considered azonal in the sense described by Mucina *et al.* (2006a) (See Section 1.4.4). This was clearly demonstrated in the differences shown to exist between wetlands associated with each of the terrestrial vegetation units within the South-West Fynbos bioregion. Further examination of climatic and geological differences as being valid drivers of these differences is explored in Chapter 4.

iii. National Freshwater Ecosystem Priority Areas

Greater significant difference was evident among the vegetation units (associated with each terrestrial vegetation unit) within the *South-west Sand Fynbos* NFEPA vegetation group than between these units and those of the *North-west Sand Fynbos* vegetation group (Table 3.7). The NFEPA vegetation groups thus incorporate too great a range of internal variability for phytoassessment purposes. This reiterates the suggestion that the terrestrial vegetation units (Rebelo *et al.* 2006) would provide phytogeographical regions with the least natural variability and thus the greatest accuracy for phytoassessment development.

iv. Zonal interface of terrestrial and wetland vegetation

Examination of the differences between the plant assemblages of wetlands as grouped by their association with different terrestrial vegetation units suggests that edaphic and climatic differences do influence wetland species geographical distributions, thus confirming the likely influence of these zonal drivers on wetland vegetation distribution. As discussed above and evident from the ordinations in figures 3.14 to 3.16, the wetland vegetation associated with each terrestrial vegetation unit are all significantly different. This confirms that similar environmental constraints would appear to act on both wetland and terrestrial units of vegetation.

The wetlands associated with the Cape Flats Dune Strandveld and Cape Flats Sand Fynbos terrestrial units, the most comprehensively sampled of any of the units of vegetation in the present study, show significantly different groupings as based on floristic assemblage. These wetlands represent not only a difference between edaphic properties in the soil but a climatic difference as they are from, respectively, the West Strandveld and the South-West Fynbos bioregions (as seen in Figures 3.10 and 3.11). Examination of whether the edaphic or the climatic differences correlate more closely with the community structure of each of these units of wetlands will be examined using correlation analyses in Chapter 4. It is apparent from the above results that, for the Cape coastal lowlands, the geographical units of wetlands associated with the zonal terrestrial vegetation units provide the most appropriate geographical units within which to attempt to develop phytoassessment tools.

3.4.4 Biotic exchange and homogenization

Even the minimally impacted wetlands of the present study contained a considerable number of alien invasive species, suggesting that it is unlikely that un-invaded wetlands still exist in the Cape coastal lowlands. It is thus not possible to compare the native diversity in un-invaded relative to that in invaded wetlands in order to ascertain if species homogenization has occurred. Result of this thesis establish that a greater percentage cover of alien species exists in impaired than in reference wetlands in the Cape coastal lowlands (Figure 3.6), similar results have been reported for rivers (Planty-Tibacchi 1996, Richardson *et al.* 2007). It is also apparent that increasing number and cover of alien relative to indigenous plant species represents a potentially useful measure of human disturbance (Figures 3.17 to 3.19). The greater strength of the coefficient of determination of the ratio of number or cover of indigenous to alien species relative to HDS than that of either the number or cover of alien or indigenous taxa with HDS suggests that the ratio is more reliable than either of its components as a measure of impairment of wetland environmental condition. These ratios suggest potential metrics of disturbance for all wetlands such that if the number or cover of aliens represents more than 30% of that of indigenous vegetation then a wetland can be considered impaired. After the removal of a pollution source or similar disturbance some inertia or time lag in recovery of the status quo of the previous environmental state and also thus of the vegetation can be expected and hence the presence of alien vegetation may not indicate an active external disturbance (Helm et al. 2006, Richardson *et al.* 2007). Relative to

indigenous herbaceous and graminoid taxa that predominate in wetlands of the lowlands of the south-western Cape (Figure 3.5), the higher fuel load (Van Wilgen & Richardson 1985), higher water, and the nutrient impact of woody alien vegetation (Witkowski & Mitchell 1987) do, however, constitute disturbances that at 30% of total cover can be considered as active negative impacts within the wetland itself.

Although the number of Cape Flats wetlands used to compare reference and impaired conditions is small, the Monte Carlo determination of significance suggests robust and significantly greater floristic variation within impaired than in reference wetlands. The analyses of similarity of reference and impaired wetlands suggest the existence of greater similarities between impaired wetlands from different vegetation units than between reference wetlands from different vegetation units (Figure 3.20). Alien species are displacing (or have displaced) a number of indigenous species and reduce the cover of other indigenous plants in wetlands of the Cape coastal lowlands. Disturbance is therefore homogenizing floristic assemblages, reducing the differences that naturally exist between minimally impaired (reference) wetlands from different vegetation units (i.e., those associated with different terrestrial vegetation units). The same tests cannot be performed with any rigour on wetlands associated with any of the other terrestrial vegetation units as too few replicates were sampled in each case. Less than five wetland replicates were sampled associated with any of the 11 other terrestrial vegetation units (See Table 2.2) as opposed to the 15 and 14 wetlands sampled associated with the Cape Flats Dune Strandveld and Cape Flats Sand Fynbos units respectively.

3.5 Chapter conclusions

There is considerable gamma diversity among wetlands of the Cape coastal lowlands as shown by the large numbers of species actually found and the larger richness anticipated to exist with the use of richness estimators and rarefaction curves. Wetlands represent islands of anomalous habitat relative to that of the surrounding terrestrial vegetation. The wetland habitat holds a considerable concentration of floristic alpha diversity within spatially much larger terrestrial units of vegetation. Considerable difference is apparent between the species assemblages of each of the West Coast, Cape Flats and Overberg sub-regions of the mediterranean region of the Western Coastal Slope. Further significant differences are apparent between the assemblages of each of the different bioregions, suggesting that climatic and geological differences on which the bioregions were based do influence wetland species distribution. Examination of the wetland

vegetation associated with each of the terrestrial vegetation units (Section 3.3.3.iv) further suggests that the climatic and edaphic characteristics that determine differences between terrestrial units also influence the geographical distribution of wetland species. Insufficient wetlands were sampled in association with many of the different terrestrial vegetation units to show significant difference between the wetlands of every unit with any certainty. Considerable beta diversity or variation in wetland plant distribution is, however, apparent along the geographical gradients of distance, climate and geology. That wetland vegetation is azonal (Walter 1973), as based for Cape Lowland Freshwater wetlands on differences brought about by hydroregime from the surrounding ambient environmental conditions (Mucina *et al.* 2006a), would appear to be inaccurate in the Cape coastal lowlands. The phytogeography of wetlands is aligned with zonal units of terrestrial vegetation that were separated with climatic and geological (edaphic) discriminators. Whilst wetland vegetation is distinct from the surrounding vegetation as a result of hydrological differences, the phytogeographical distribution of wetland vegetation cannot be considered azonal in the sense that Mucina *et al.* (2006a) describe in which hydroregime was considered to "*exert an influence greater than the macroclimate on floristic composition*". The importance of these findings will be discussed in the conclusions to the thesis. It is important to note that significantly different units of wetland vegetation exist within the Cape Lowland Freshwater vegetation unit that was mapped by Mucina *et al.* (2006a) as a single wetland vegetation unit occurring across the whole of the Cape coastal lowlands including the wetland regions of both the mediterranean Western Coastal Slope (SW.m) and of the temperate Southern Coastal Slope (SS.a) (*sensu* Cowan 1995). The freshwater vegetation of the Cape coastal lowlands would perhaps more accurately be considered a "metacommunity," which is a very large collection of similar organisms found across a biogeographic region (Hubbell 2001). What constitutes different wetland phytogeographic regions in the Cape coastal lowlands might alternatively be considered to be each of the western, central and eastern sub-regions of the present study, Cowan's (1995) wetland regions or the proposed NFEPA wetland vegetation groups (Roux *et al*. 2006) that combine climatically and geologically related groups of wetlands. Further baseline data is required to be able to determine:

i. If Cowan's (1995) mediterranean Western Coastal Slopes region (SW.m) in its entirety constitutes a different biogeographic region from the temperate Southern Coastal Slopes region (SS.a); and
ii. If wetland vegetation associated with Hopefield and Atlantis Sand Fynbos vegetation units of the NFEPA South West Sand Fynbos group are

consistently more similar to the North West Sand Fynbos incorporating Leipoldtville Sand Fynbos than to the Cape Flats Sand Fynbos.

This was beyond the scope of the data collated for the present study. For purposes of phytoassessment development, recognition of different phytogeographic units of wetland vegetation is necessary in order to assist in the detection of floristic difference brought about by human disturbance as opposed to by natural environmental differences. The terrestrial vegetation units (Rebelo *et al.* 2006) represent phytogeographically associated groups of wetlands with less natural interwetland variability due to climatic and geological variation than any of the previously addressed units of phytogeography. Classifying the wetlands associated with each of the terrestrial units as different regions for purposes of phytoassessment development introduces a large number of subdivisions but considerably reduces natural variability within each region relative to that encompassed by previously discussed regional units.

The impacts of human disturbance on species distribution can be seen to reduce floristic differences between units of distinct wetland vegetation as associated with different terrestrial vegetation units (*sensu Rebelo et al.* 2006) as was shown in Figure 3.20. Furthermore a greater ratio of the number and cover of alien relative to indigenous species was evident in the impaired than in the reference (minimally impaired) wetlands of the Cape coastal lowlands. Human influences can thus be seen to change wetland floristic composition, suggesting potential for development of phytoassessment related tools. Development of phytoassessment methods for the Cape coastal lowlands will be explored in Chapter 5. The influence of edaphic characteristics, climate, hydroregime and human disturbance on wetland species distribution is addressed in the following chapter.

CHAPTER 4

DIFFERENCES IN COMMUNITY STRUCTURE IN RELATION TO SPATIAL AND ENVIRONMENTAL VARIABLES

4.1 Introduction

The objective of this chapter is an examination of the relationship between the distribution of wetland macrophytes and the spatial, climatic, geological and anthropogenic variables that are presumed to be the drivers of wetland phytogeography within the Cape coastal lowlands. Such information can guide the classification of regions anticipated to have homogenous wetland vegetation under minimally impaired conditions. Although significant correlations between spatial and environmental variables and floristic communities do not signify causal effects underlying observed ecological patterns, the strength of correlations suggests whether these variables are likely to be at least partially responsible for the observed patterns. In such path analyses, the lack of correlation between two variables argues against a causal relationship (Urban 2003). Knowing which environmental parameters have the strongest relationships with floristic community patterns guides our ability to interpret the impacts of anthropogenic and natural changes over space and time and guides further autecological exploration. Ascertaining which environmental variables are most strongly correlated with spatial differences is useful for establishing how each unit of vegetation should be conserved (Mucina *et al.* 2006a).

The analyses reported in Chapter 3 suggest inland freshwater seasonal wetlands of the Cape coastal lowlands do not support a phytogeographically homogenous unit of vegetation. Climatic and edaphic factors were hypothesized to impact on the geographical distribution of species. The importance of these factors was inferred from geographical discontinuities between wetland plant communities from different bioregions and between terrestrial vegetation units that are associated with different edaphic substrates. Whilst the bioregions stretch across sub-regions, the terrestrial units are subsets of the bioregions or of the sub-regions; hence each terrestrial unit represents a spatial subset of the larger bioregions and sub-regions. Whether climatic or edaphic factors have a significant relationship with these differences in floristic community structure will now be addressed.

The goal of explaining the distribution of species in terms of environmental variables presumed to be the operative constraints on the species is confounded by two fundamental issues. Firstly environmental variables are intercorrelated (collinear) among themselves and it is thus difficult to imply a cause and effect mechanism to a variable even if it correlates with species distribution. Secondly, it is not easy to separate the effect of environment and space in regard to their impact on the distribution of species, because much of the variance attributed to environmental variation can alternatively be explained by the spatial pattern underlying it (Legendre 1993). This interrelationship of space and environmental data (such as the change in climate over an area) is defined as the lack of independence between pairs of observations at a given distance apart and is known as spatial autocorrelation (Legendre & Legendre 2003). Spatial autocorrelation refers specifically to the lack of independence among the error components of pairs of observations of field data, due to spatial proximity (Cliff & Ord 1981, Legendre 1993, Legendre & Legendre 2003). In addition the species themselves may exhibit patchiness in distribution due to biotic processes such as dispersal, thus implying the existence of spatial autocorrelation in distribution (Legendre & Fortin 1989, Urban 2003). The Mantel test (Mantel 1967), and derived forms thereof, is an approach that overcomes some of the problems inherent in explaining species-environment relationships. The Mantel test shows the interrelationship of the entire set of environmental variables (or subsets thereof) to the species variables and is thus useful in determining plausible paths for cause and effect. A portion of information is unexplained by Mantel tests, this residual may be the result of unmeasured environmental variables, or of biotic interaction such as interspecific competition, mutualism, or dispersal.

Canonical analysis is a different approach to explaining species-environment relationships that has the ability to highlight individual environmental variables with the strongest correlations to species distribution. Canonical analysis of principal coordinates (CAP) sphericises all variables (see method description below), making them all comparable, thereby facilitating the identification of variables with the strongest correlations to the observed distribution of species within a data set (Anderson & Willis 2003). Canonical analysis has greater statistical power to show a relationship when one is present in the data than the Mantel test (Legendre & Fortin 2010). In addition, Mantel tests underestimate the proportion of the original data variation explained by spatial structures (Legendre & Fortin 2010). Both Mantel tests and canonical analysis were employed in the present study as their combination allows interpretation of sets of variables and of individual variables within the set. This chapter uses Mantel tests to search for

significant correlations between sets of variables that may explain some of the phytogeographical patterns observed to exist in the wetland vegetation of the Cape coastal lowlands. Thereafter, CAP is used to determine which environmental variables are most strongly correlated with the observed phytogeographical patterns.

4.2 Methods for explaining species-environment relationship

The spatial variables measured for each wetland were latitude, longitude and altitude. The edaphic variables consist of a combination of the physical and chemical variables that describe the substrate in each wetland, as described in Section 2.2.61. The climatic variables consist of mean daily minimum and maximum temperatures as well as mean annual rainfall and evaporation (Section 2.2.7 (i)). The disturbance variables consist of the aspects used in the development of the Human Disturbance Score, namely physical impacts, water quality impacts, hydrological impacts and the extent of buffer width as well as the extent of woody alien vegetation cover (surrounding the wetlands) and of vegetation utilization within wetlands (Section 2.2.7 (iii)). All environmental variables including the edaphic and water physico-chemical properties of the wetlands and the climatic and anthropogenic data for the surrounding area are in the appended CD (Appendix 6).

The water column physico-chemical variables were not taken into consideration for the purposes of this exploration as a considerable amount of missing data existed within this data set. Furthermore the anticipated variation over diel, weekly and seasonal time scales, particularly as a result of hydroperiod in seasonal wetlands suggest that these spot measurements represent transient values (pers. com. Assoc. Prof's. Jenny Day & Mike Lucas, UCT). Only variation at seasonal time scales is likely to affect nutrient concentrations in the soils (Witkowski & Mitchell 1987, Mitchell *et al*. 1987) hence these variables were considered of greater value than the water column variables for interwetland comparison and autecological development.

4.2.1 Mantel tests at the biome scale

Mantel tests were used to identify correlations between environmental data (and subsets of this data) with floristic community structure of Cape coastal lowland wetlands of the Fynbos biome. The Mantel test (Mantel 1967), is a statistical test of the correlation between two resemblance matrices and reflects the interrelations between vectors of paired field measurements (e.g.

floristic community and environmental data) of a set of objects (e.g. wetlands). The test is commonly used in ecology, where the data are estimates of the resemblance between objects such as wetlands (Sokal & Rohlf 1995). The resemblance is either a measure of distance or similarity where dissimilarity is a measure of ecological distance and similarity the additive inverse of dissimilarity. Whether a resemblance matrix reflects distance or similarity depends on the resemblance coefficient chosen (see Legendre & Legendre 2003). As dissimilarity (*D*, ecological distance) typically is equivalent to the additive inverse of similarity (*S*) (i.e.: *D=1-S*), using similarity (or closeness) instead of dissimilarity (distance) has no qualitative effect on the analysis: it merely changes the sign of the coefficients and thus also of the outcome of analysis based on the resemblance matrix vectors. The results of the Mantel tests are therefore reported only as positive correlations. The Mantel test performs a regression of the distances (dissimilarities or similarities) represented in the resemblance matrices after, in the most accurate approach, first permuting only one of these matrices (Legendre 2000). The relationships between macrophytes, environment and spatial distribution were assessed using the Mantel test (Mantel 1967). For this analysis, all of the above listed edaphic, climatic and anthropogenic environmental variables were combined into a single "environmental" resemblance matrix. The Mantel statistic (r_M) was calculated for each matrix pair (e.g. species *vs* environment, environment *vs* space and species *vs* space); and tested to see if it differed from the expected null hypothesis of no correlation using Monte Carlo permutations tests (999 iterations).

An extension of the Mantel test is to perform a partial Mantel test to account for the relative proportion of community structure explained by the different sets of variables (e.g. environment and spatial). In this test the third (or more) matrix is held constant (thereby first accounting for the variation within this matrix) while the relationship of the first two is determined (Smouse *et al.* 1986). The adequacy of the partial Mantel test as a test procedure was comprehensively reviewed by Legendre (2000). The validity of the permutation of only the species variables before regression of the paired matrices was recently recommended as the best approach for tests of significance for the partial Mantel test by Legendre & Fortin (2010). Adopting this approach, studies with a large number of replicates ($n \geq 50$), with skew or normal error distribution, and with or without outliers in the covariables (those variables held constant during the analysis), and without outliers in the species matrix will return accurate estimates of the significance for partial Mantel statistics (Legendre 2000, Legendre & Fortin 2010). Mantel and Partial Mantel tests were done in *PASSaGE 2* (Rosenberg & Anderson 2011) a program for

spatial analysis using the approach recommended by Legendre (2000) and Legendre & Fortin (2010). The absence of outliers in the species matrix is evident from the ordination of wetlands in Chapter 3 (e.g. Figures 3.7 & 3.9). The interrelationship of environment, macrophytes, and space were thus also assessed using the partial Mantel test (Smouse *et al.* 1986, following the procedural recommendations of Legendre (2000)) by holding constant the third matrix during the regression of any of the other two resemblance matrices. Furthermore a multiple partial Mantel test was performed by splitting the environmental variables into sets based on edaphic, climatic and anthropogenic influences. In this process during the comparison of any pair of resemblance matrices (e.g.: species *vs* space) all of the other three matrices (e.g.: climatic, edaphic and anthropogenic) were held constant. Tests of significance were again made using Monte Carlo permutations (999 permutations).

4.2.2 Canonical analysis at the biome scale

The correlation of individual environmental variables with the community structure of Cape coastal lowland wetlands of the Fynbos biome were examined using canonical analysis. Canonical correlation of the linear combination of the environmental variables (including spatial variables) and the linear combination of the species cover values in the space of the principal coordinate axes generated from each set of variables provides a means of exploring the correlational relationships between the two sets of variables (Anderson & Willis 2003). Canonical Analysis of Principle coordinates (CAP) was performed with the PERMANOVA add-on to PRIMER-E (Anderson *et al.* 2008). In the same manner as the traditional Canonical Correlation Analysis (or CCA (ter Braak 1990)), CAP sphericises both the environmental and species data before relating them to one another, thus performing a multivariate normalization of the data, standardizing the dispersions of individual variables and removing the correlation structure among the variables (Anderson *et al.* 2008). Thus each variable has a mean of zero a length of one and a correlation of zero and is directly comparable to every other variable. The linear combination of sphericised environmental variables that best explain the resemblances of sphericised community data are projected onto a graph as potential driving environmental vectors most correlated with the ordination of the community resemblance. The length and direction of a vector representing an individual environmental variable correspond to the strength and the direction of influence of that variable. This exploration is used in the present study to ascertain which environmental variables have the strongest correlation to phytogeographical differences between spatially disparate units of vegetation.

The CAP procedure is susceptible to skewness and multi-collinearity in the environmental variables as well as to outliers from either the environmental or species data sets (Anderson & Willis 2003). To ensure that all the environmental variables have reasonably symmetric distributions and even scatter it is necessary to subject variables with skew distributions to ($\log_e$) transformation (Clarke & Warwick 2001) before performing the CAP. A search for any multi-collinearity between variables in the environmental data set was performed using Pearson correlations (*r*). One variable from any pair of variables that were collinear at ±90% ($r = \pm 0.90$) were removed from the data set as they were considered to contain virtually the same information and are redundant for purposes of the analysis. Given the differences shown to exist between the wetland vegetation of each sub-region (West Coast, Cape Flats and Overberg) in Chapter 3, potential drivers of difference between the West Coast and Cape Flats data were explored independently from an examination of the Cape Flats and Overberg data. An ordination of the environmental variables, other than the spatial variables of latitude, longitude and altitude, was performed to examine the environmental differences between the sub-regions. The spatial variables were specifically excluded as by the nature of geographical distances these variables would have a predictable spatial influence between sub-regions that is not of interest in an ordination performed to search for difference between the environmental parameters of sub-regions. The existence of considerable environmental differences between sub-regions would further support the decision to search for variables correlated with such differences by performing CAP separately for West Coast *vs* Cape Flats and Cape Flats *vs* Overberg.

4.2.3 Species–environment relationship for Cape Flats wetlands

At finer spatial scale than the differences across the Fynbos biome discussed above, species, spatial and environmental relationships were examined within the central sub-region of the Fynbos Biome on the Cape Flats. The combination of wetlands associated with the different edaphic substrates of the terrestrial Cape Flats Dune Strandveld and Cape Flats Sand Fynbos vegetation units provides a dataset within which sufficient replicate wetlands were sampled (n = 15 and 14) in order to be representative of the macrophytic diversity of the Cape Flats (Section 3.4.1). This sub-region scale thus provides an opportunity to ascertain the impact of climatic, edaphic and anthropogenic differences between wetlands associated with these different terrestrial vegetation units. In Chapter 3 it was shown that:

i. There was a significant difference between the community structure of all wetlands from the West Strandveld and South West Fynbos bioregions (which represent areas with different climatic and geological characteristics: Figure 3.10 and 3.11); and
ii. That there was a significant difference between the community structure of the wetlands associated with the (intra)zonal terrestrial vegetation units of Cape Flats Dune Strandveld and Cape Flats Sand Fynbos (each of which is associated with a different edaphic unit (calcareous & alkaline *vs* acidic sands): Figure 3.15, as well as the different climates as associated with different bioregions).

At this sub-regional scale the interrelationship of edaphic, climatic, anthropogenic and spatial variables were examined in order to ascertain the strongest correlations to species distribution patterns within the wetland vegetation associated with terrestrial Fynbos and Strandveld. Mantel tests and partial Mantel tests and a CAP were performed to ascertain whether climatic or edaphic differences were more strongly correlated with observed community structure differences between Fynbos- and Strandveld-associated wetlands. This information would guide whether to focus phytoassessment development efforts within sub-regions, within bioregions, or within the edaphically discriminated and (intra)zonal terrestrial vegetation units within the sub- or bioregions.

4.3 Results

4.3.1 Correlation of environmental to species resemblances at the biome scale

At this biome-wide scale significant correlations (Mantel 1967) and partial mantel correlations (Smouse *et al.* 1986) are apparent between species, environmental and spatial variables (Table 4.1). Results provided above the diagonal are for Mantel tests and those below the diagonal for the partial Mantel tests. When keeping spatial variables constant (i.e. using partial Mantel tests and first dealing with the variation explained by spatial variation), environmental variables in this study had a 30% correlation with the variation in wetland macrophyte assemblages in the Cape coastal lowlands, leaving 70% of the correlation unexplained. Spatial variables accounted for almost a quarter of the observed macrophyte variation (a correlation of 24%) when environmental variables were kept constant.

A second partial Mantel test using multiple regression of separate sets of the spatial, climatic, edaphic, and anthropogenic variables along with the resemblance of the floristic community

structure of wetlands of the Cape coastal lowlands was also performed. Separation of the environmental variables into edaphic, climatic and anthropogenic (disturbance) influences suggested that the climatic variables had the strongest correlation with the variation in species distribution across the Cape coastal lowlands accounting for 24% of the variation when disturbance and edaphic variables were kept constant (Table 4.2). The edaphic and spatial variables had 15% and 14% correlation with the variation in macrophyte distribution in partial Mantel tests. Biotic interaction and un-measured environmental variables are considered important in explaining some of the residual of unexplained variation of the species assemblage. After this separation of the environmental variables into components, anthropogenic disturbance was not significantly correlated to other variables in simple or partial mantel tests. Space and geology also show insignificant partial correlation when species, disturbance and climatic data are held constant. Anthropogenic disturbance was thus not significantly correlated with the distribution of macrophytes when examined at the biome-scale for the Cape coastal lowlands.

Table 4.1: Mantel and partial Mantel test (r_M) between spatial, environmental and species variables for the wetlands of the Cape coastal lowlands. Above the diagonal are simple Mantel test results, below the diagonal are partial Mantel tests. Significance based on 999 permutations of two tailed tests is shown with ** indicative of p = 0.001.

	Species	Environment	Space
Species		0.43 **	0.39 **
Environment	0.31 **		0.46 **
Space	0.24 **	0.35 **	

Table 4.2: Mantel and partial Mantel test (r_M) between spatial, edaphic, climatic, anthropogenic disturbance and species variables for the wetlands of the Cape coastal lowlands. Above the diagonal are simple Mantel test results, below the diagonal are partial Mantel tests. Significance based on 999 permutations of two tailed tests is shown with *indicative of p = 0.01 and ** indicative of p = 0.001.

	Species	Edaphic	Climatic	Disturbance	Space
Species		0.29 **	0.46 **	0.05	0.39 **
Edaphic	0.15 *		0.38 **	0.01	0.21 *
Climatic	0.24 **	0.27 *		0.003	0.66 **
Disturbance	0.03	0.02	0.13		0.14
Space	0.14 *	0.07	0.59 **	0.19*	

4.3.2 Canonical analysis at the biome scale

Examination of the collinearity of the environmental variables revealed that there were four pairs of highly collinear variables as displayed in Table 4.3. One of each pair of collinear variables (the "redundant" variables in Table 4.3) was removed from the environmental data before performing an ordination and canonical analyses.

Table 4.3: Collinear environmental variables as determined with Pearson correlation in PRIMER-E

Retained variables	Pearsons *r* (correlation)	Redundant variables
K	0.98	$K_{\text{exchangeable cations}}$
$Na_{\text{water soluble}}$	0.92	$Na_{\text{exchangeable cations}}$
$K_{\text{water soluble}}$	0.93	$Mg_{\text{water soluble}}$
% Clay	-0.96	% Sand

After removal of the collinear variables an ordination of the environmental variables of all three sub-regions was performed as illustrated in Figure 4.1.

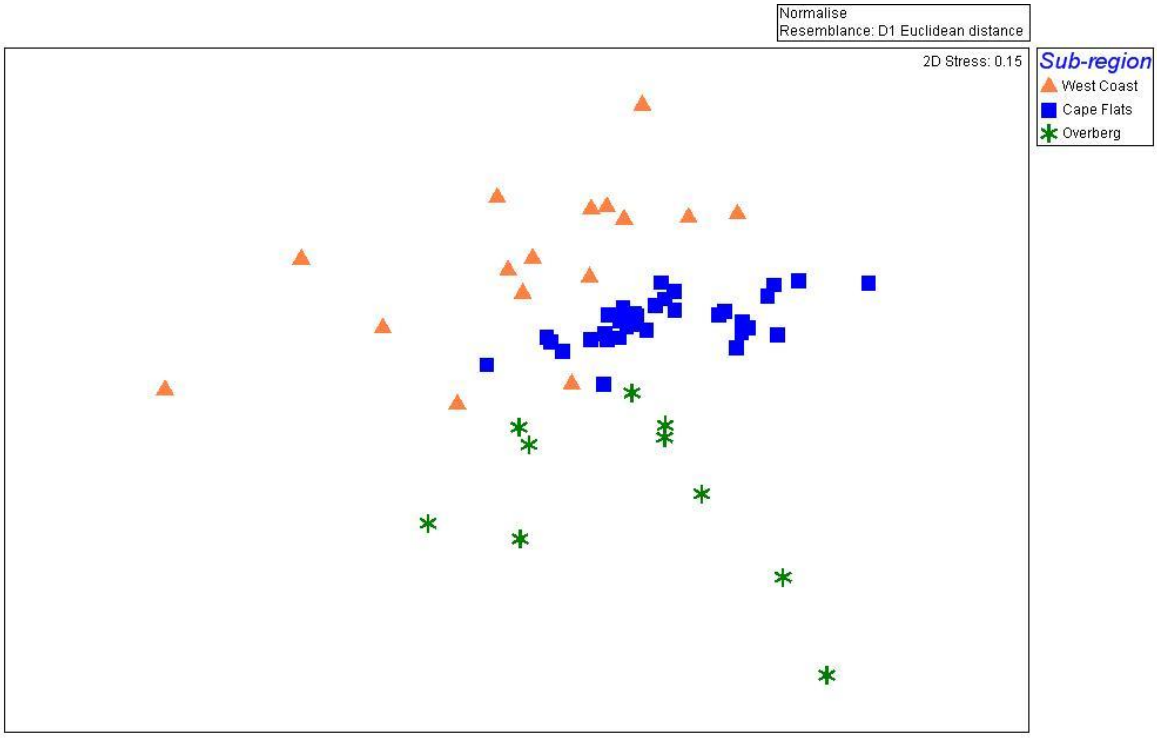


Figure 4.1: Non-metric multidimensional scaling ordination of the wetlands of the Cape coastal lowlands showing their relationship based upon environmental variables that were collinear at less than ±90% and excluding the spatial variables of latitude, longitude and altitude.

Clear distinction is apparent between the West Coast, Cape Flats and the Overberg wetlands that were sampled as being representative of the western, central and eastern sub-regions of the Cape coastal lowlands. In order to search for environmental variables correlated with community differences between sub-regions, Canonical Analysis of Principle Coordinates (CAP) was performed separately for the combined West Coast and Cape Flats data and for the combined Cape Flats and Overberg data. The results of these analyses are displayed in Figures 4.2 and 4.3 for the West Coast and Overberg respectively.

i. West Coast *vs* Cape Flats

The length and direction (relative to both CAP axes and grouped sub-region samples) of the vectors of rainfall and evaporation suggest these variables are best at explaining differences between the community structure of the West Coast and Cape Flats sub-regions. As displayed in Figure 4.2 (and in Table 4.4), evaporation (greater on the West Coast than on the Cape Flats) and rainfall (less on the West Coast than on the Cape Flats) have the highest correlations, of all measured environmental variables, with the floristic community structure. A generally greater percentage of clay and lower percentage of soil carbon recorded in the West Coast than Cape Flats samples, as well as higher levels of human disturbance, also have strong correlations with species patterns and thus potentially influence the observed differences in the floristic community structure of the West Coast and Cape Flats wetlands. The direction of the vectors of the clay, carbon and HDS variables suggests that they are less correlated with (important in driving) difference between the west and central sub-regions as among sites within each of them. Table 4.4 provides a summary of the mean value per sub-region of these five environmental variables. The Overberg values and other variables with strong correlations with the Cape Flats and Overberg community structure are also displayed in Table 4.4 for comparative purposes.

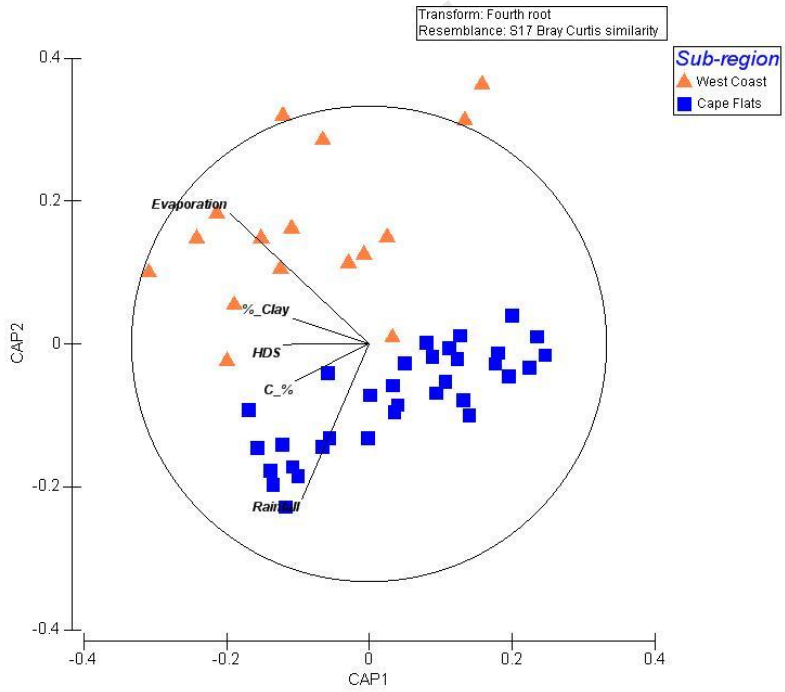


Figure 4.2: Canonical Analysis of Principal Coordinates of the environmental and species data sets of the West Coast and Cape Flats wetlands of the Cape coastal lowlands. The overlay vectors are of the environmental parameters that correlate highest (greater than 30%) with the principal coordinate axes. These vectors overlay an ordination of the sphericized species cover resemblances of the community structure of each wetland.

Table 4.4: Mean values (± Standard Error) for variables with strong correlations (>30%) to the distribution of the wetland macrophyte assemblage of the West Coast, Cape Flats and Overberg areas in the Cape coastal lowlands.

Variable:	Evaporation	Rainfall	% Clay	% Carbon	H^+_{exca}**	Ca_{exca}	$Ca_{water\ soluble}$	HDS*
(Unit)	(mm)	(mm)	(%)	(%)	($cmol_c.kg^{-1}$)	($cmol_c.kg^{-1}$)	($cmol_c.kg^{-1}$)	(score)
West Coast	2299 (±20)	326 (±23)	11.5 (±3)	0.95 (±0.2)	0.26 (±0.1)	4.5 (±1)	60 (±18)	120 (±16)
Cape Flats	1991 (7)	765 (±39)	0.78 (±0.1)	1.2 (±0.07)	0.27 (±0.08)	9.5 (±1)	70 (±9)	96 (±6)
Overberg	1830 (20)	561 (±7)	9.9 (±4.8)	2.5 (±0.8)	1.6 (±0.8)	6.1 (±2)	57 (±20)	76 (±9)

** H^+ exca = exchangeable cations of hydrogen

*HDS = Human Disturbance Score

ii. Cape Flats *vs* Overberg

Between the Cape Flats and Overberg wetlands, of all environmental variables contrasted by canonical analysis, evaporation has the closest correlation with the observed variation in community structure (Figure 4.3) Calcium content of the soil as measured by both the amount of water-soluble calcium and to a lesser extent (as apparent from the direction and vector length) exchangeable cations of calcium, also had greater than 30% correlation with the first two principal coordinate axes, thus explaining some of the observed phytogeographical pattern of the macrophytic community structure on the Overberg and Cape Flats. Clay, human disturbance and the exchangeable cations of hydrogen also have strong correlations with the phytogeographical pattern of the overall community structure but the direction of their vectors suggests less influence on the observed differences between sub-regions than on difference within sub-regions. A summary of the mean value per sub-region of these environmental variables are presented in Table 4.4 for comparative purposes. On the Cape Flats a difference between the Sand Fynbos and Dune Strandveld samples is apparent in Figure 4.3 as indicated by the groups of wetlands enclosed within the dashed and solid ellipses.

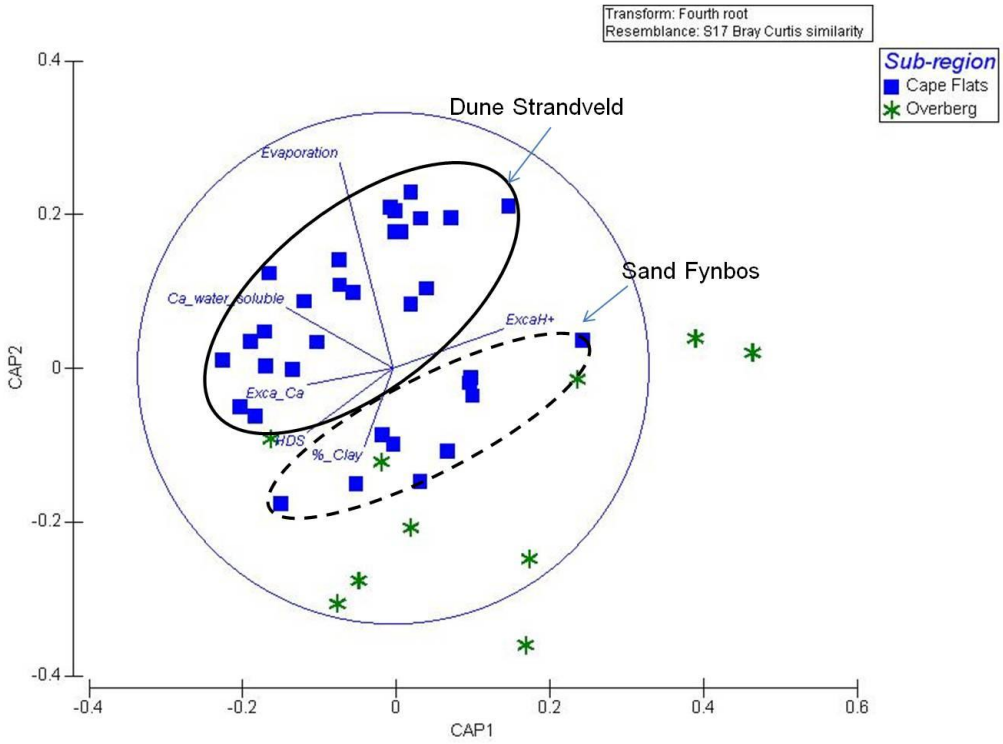


Figure 4.3: Canonical Analysis of Principal Coordinates of the environmental and species data sets of the Cape Flats and Overberg wetlands of the Cape coastal lowlands. The overlay vectors are of the environmental variables that correlate most (greater than 30%) with the principal coordinate axes. These vectors overlay an ordination of the sphericized species cover resemblances of the community structure of each wetland. The solid and dashed ellipses respectively encircle the Cape Flats Dune Strandveld and the Cape Flats Sand Fynbos wetlands.

4.3.3 Species–environment relationship for Cape Flats wetlands

i. Mantel tests

For the Cape Flats data partial Mantel tests suggest considerable correlation between environmental variables and species but not between environmental variables and space as displayed in Table 4.5. Using the simple Mantel test the human disturbance score was not significantly correlated with community structure on the Cape Flats. Collectively the combination of the four factors making up the HDS (water quality, hydrological & physical disturbances, & buffer loss), the extent of woody alien vegetation in the first 100 and following 400 meters around wetlands and the amount of vegetation utilization were significantly correlated to the community structure of Cape Flats wetland vegetation (r_M = 0.21, p = 0.001) using simple Mantel tests (Table 4.6). According to the partial Mantel test reported in Table 4.6, after controlling for the correlation of climatic, edaphic and spatial variables disturbance was no longer significantly correlated with community structure. In contrast to the partial Mantel test of environment *vs* space, multiple-partial-Mantel tests of the subsets of this environmental data

reveal that climatic variables have the strongest correlation with space, geology and species after controlling for the variation of other variables.

Table 4.5: Mantel and partial Mantel test (r_M) between spatial, environmental and species variables for the wetlands of the Cape Flats. Above the diagonal are simple Mantel test results, below the diagonal are partial Mantel tests. Significance based on 999 permutations of two tailed tests is shown with *indicative of p = 0.01 and ** indicative of p = 0.001.

	Species	Environment	Space
Species		0.45 **	0.23 **
Environment	0.42 **		0.19 *
Space	0.16 *	0.1	

Table 4.6: Mantel and partial Mantel test (r_M) between spatial, edaphic, climatic, anthropogenic disturbance and species variables for the wetlands of the Cape Flats. Significance based on 999 permutations of two tailed tests is shown with *indicative of p = 0.01 and ** indicative of p = 0.001.

	Species	Edaphic	Climatic	Disturbance	Space
Species		0.35 **	0.52 **	0.21 *	0.22 **
Edaphic	0.09		0.55 **	0.20 *	0.09
Climatic	0.37 **	0.46 **		0.25**	0.35 **
Disturbance	0.09	0.085	0.08		0.18 *
Space	0.05	0.07 *	0.3 **	0.11	

ii. Canonical analysis

Due to the insignificant correlation of HDS shown with the Mantel tests, this variable was replaced by its component parts (water quality, hydrology, physical disturbance and buffer width extent) and the spatial extent of alien vegetation cover surrounding wetlands and the amount of vegetation utilization. Examination of the collinearity of environmental variables, excluding spatial variables, revealed eight pairs of highly collinear variables, as displayed in Table 4.7. The redundant variable from each pair (as indicated in Table 4.7) were removed from the environmental dataset before running the CAP that is presented in Figure 4.4.

Table 4.7: Collinear environmental variables as determined with Pearson correlation

Retained variables	**Pearsons *r* (correlation)**	**Redundant variables**
pH soil	0.95	$Ca_{\text{exchangeable cations}}$
pH soil	-0.93	Rainfall
Soil conductivity	0.97	$Na_{\text{water soluble}}$
Soil conductivity	0.95	$Mg_{\text{water soluble}}$
K	0.92	$K_{\text{water soluble}}$
$K_{\text{water soluble}}$	0.9	$Na_{\text{water soluble}}$
% woody alien vegetation in 100m	0.93	% woody alien vegetation in 500m
Evaporation	-0.91	$H^+_{\text{exchangeable cations}}$

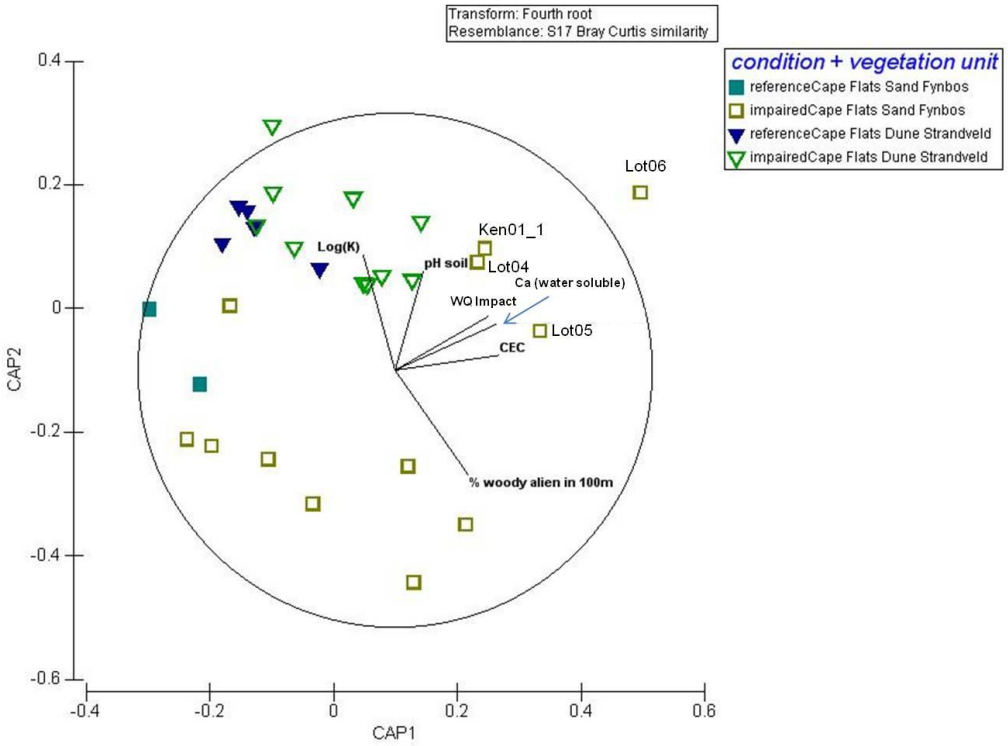


Figure 4.4: Canonical Analysis of Principal Coordinates of the environmental and species data sets of the Cape Flats Dune Strandveld and the Cape Flats Sand Fynbos wetlands. The overlay vectors are of the environmental variables that correlate most strongly (greater than 40%) with the principal coordinate axes. These vectors overlay an ordination of the sphericized species cover resemblances of the community structure of each wetland.

The length and direction (relative to both CAP axes and grouped vegetation samples) of the vectors of [K_{loge}] (log(K) in Figure 4.4), of soil pH, water quality impact, water soluble calcium and cation exchange capacity (CEC) (and their related collinear variables in Table 4.7) suggests that these variables have strong correlations with the community structure of these Cape Flats wetlands. These variables specifically correlate to the distribution and difference separating most of the Fynbos wetlands from the combination of the Fynbos-associated-wetlands Lot04, Lot05, Lot06 and Ken01_1 and the Strandveld wetlands on the Cape Flats. The length and orientation of the vector of percentage woody alien vegetation cover and [K_{loge}] suggest that these variables have a significant relationship with the difference between reference and impacted wetlands of both vegetation units. The mean values of each of these vectors and of their collinear variables are presented in Table 4.8.

Table 4.8: Mean values (± Standard Error) for environmental variables with strong correlations (>40%) to the distribution of the macrophyte assemblage of the Fynbos and Strandveld wetlands on the Cape Flats

Variable:	pH	K	CEC	$Ca_{water\ soluble}$	WQ Impact	Rainfall	% alien woody 100	% alien woody 500
(Unit)	pH units	($mg.kg^{-1}$)	($cmol_c.kg^{-1}$)	($cmol_c.kg^{-1}$)	(score)	(mm)	(%)	(%)
Fynbos	5.4 (±0.3)	33 (±16)	3 (±0.5)	39 (±15)	43 (±5)	987 (±30)	15 (±5)	19 (±5)
Strandveld	8.2 (±0.1)	40 (±6)	3 (±0.2)	83 (±7)	43 (±6)	553 (±17)	0	0

In contrast to the results of the Mantel tests, canonical analysis shows that, besides the climatic variable of rainfall, edaphic and disturbance related variables do have strong correlations to the distribution of wetland vegetation on the Cape Flats. Furthermore, a significant spatial gradient is evident in the CAP as shown by the separation of most Fynbos-associated and Strandveld-associated wetlands.

4.4 Discussion

Correlations, ordinations and canonical analysis of principle coordinates all suggest the biogeography of wetland macrophytes is related to the spatial arrangement of climatic and edaphic parameters. These two parameters explained 24% and 15% respectively of the biogeographical distribution of wetland macrophytes in the Cape coastal lowlands. Within single edaphic units of vegetation, correlation was also apparent between anthropogenic disturbance and macrophyte distribution, which was not the case as the biome-scale. The results achieved with these different analyses performed at the biome and sub-regional scales thus provided evidence of correlations of the measured environmental variables with the distribution of wetland macrophytes of the Cape coastal lowlands. These correlations explain significant spatial and environmental relationships with the phytogeography of wetland plants that assist in determining phytogeographical units with the least natural variation within which to develop metrics for phytoassessment.

4.4.1. Mantel correlations at the biome scale

Anthropogenic disturbance was not significantly correlated with the distribution of macrophytes when examined at the biome-scale for the Cape coastal lowlands. This lack of correlation is likely to be the result of differences in community structure caused by anthropogenic disturbance being obscured by the large amount of natural variation that exists between wetlands from the different spatial units, namely the sub-regions, bioregions and edaphic units. Within single edaphic units significant correlation does appear to exist between disturbance and

macrophyte distribution as evidenced by the ordination of the community structure of Cape Flats Dune Strandveld and Sand Fynbos wetlands in Figure 3.20.

The significant correlation of the spatial and environmental variables with the distribution of the wetland macrophytes suggests that a spatial gradient exists in the multivariate ecological data. Within this framework of a significant spatial gradient in the measured edaphic and climatic variables, based on their proximity to each other, wetlands that are furthest apart can be seen to be different whilst those close together are relatively similar. In other words ecological distance increases as the samples get to be geographically farther apart (*sensu* Legendre and Fortin 1989). Spatial autocorrelation in the patchy distribution of wetlands in geographical space may explain some of the variation in species distribution. All wetlands will have some similarity simply by the nature of being similar habitats; as well as due to successful dispersal events in the past (Santamaría 2002). These results, therefore, support the suggestion that the wetlands of the Cape Flats, West Coast and the Overberg each represent relatively distinct units of wetland vegetation and that the climatic and, to a lesser extent, the edaphic conditions in these different sub-regions are significantly correlated with their separation and their distribution.

4.4.2 Biome-scale species–environment relationship with canonical correlation

The existence of a significant gradient in the environmental data is supported by the separation of each of the sub-regions in the ordination in Figure 4.1. The results of the CAP analysis further support the above interpretation that there is a significant spatial and resultant ecological gradient within the environmental data at the biome-scale. The increasing evaporation gradient from south-east to north-west of the study area was shown by the CAP analyses to be strongly correlated to the observed phytogeographical pattern of wetland vegetation in the Cape coastal lowlands (Figures 4.2 & 4.3). In addition the strong concentration of West Coast rainfall in winter and the broader seasonality of rainfall in the Cape Flats and Overberg (Chase & Meadows 2007) would suggest that the duration of saturation of similar substrates would be lowest on the West Coast. Whilst the Cape Flats and Overberg have relatively similar moisture deficits the Overberg wetland sites have, on average, greater moisture stress than the Cape Flats sites (Table 2.1, Deacon *et al.* 1992, Schulze 2006); with more arid conditions in both summer and winter in the Overberg than the Cape Flats. Greater total evaporation rates in the Cape Flats than the Overberg wetlands (Table 2.1, Schulze 2006) do appear to have a strong correlation with and thus influence the environmental and species differences between the Cape

Flats and the Overberg wetlands as is apparent by the length and direction of the evaporation vector in Figure 4.3. The difference in moisture deficit between Strandveld and Fynbos vegetation units on the Cape Flats also appears to be influencing this strong correlation in the CAP as represented in Figure 4.3; considering the greater number of samples from either of these than from the Overberg vegetation units this is perhaps not surprising. Correlations of climatic variables to the community structure of the sub-regions are thus emphasized by CAP and similar relationships were suggested by the correlations obtained using partial Mantel tests. These analyses also both revealed a number of edaphic and anthropogenic disturbance variables with strong correlations with community structure as summed up below.

The greater clay percentage in the West Coast than Cape Flats wetlands is partially related to the fact that wetlands associated with Renosterveld and its related shale-dominated and clay-bearing granitic and alluvium sediments (Rebelo *et al.* 2006) were sampled on the West Coast whilst only sandy substrates were sampled on the Cape Flats. Two wetlands in the Overberg, both within Elim Ferricrete Fynbos, had 30 and 40% clay contents in their sediments, thereby creating considerable variability in the clay content of the Overberg and suggesting that transformation of this variable may have reduced the importance of clay in the CAP exploring differences between Cape Flats and Overberg community structure. Lack of replicate wetlands in both West Coast and Overberg, where clay substrates were most represented, reduce the potential to determine the ecological importance of this variable. Most simply though a difference in wetland communities between clay rich and clay lean poor substrates was highlighted.

Greater calcium content in Cape Flats wetland sediments than those of the Overberg and West Coast reflects the proximity of the Cape Flats wetlands to the sea and the large number of Strandveld wetlands sampled within a known calcareous rich substrate (Table 2.1). No Strandveld wetlands were sampled in the Overberg and only three were sampled on the West Coast hence calcareous rich substrates were predominantly only sampled on the Cape Flats. The length and direction of the vector for water soluble calcium suggests significant correlation with the difference between Cape Flats and Overberg wetlands as shown by the CAP in Figure 4.3, however, it is not correlated with the community structure differences between Cape Flats and West Coast wetlands in Figure 4.2.

Differences in carbon content between West Coast and Cape Flats wetlands are on average minimal and the influence (vector direction) of this variable does not appear to influence difference between sub-regions as much as differences among wetlands within sub-regions. Exchangeable cations of calcium and of hydrogen correlate with intra- rather than inter-sub-region differences in the Overberg and Cape Flats (Figure 4.3). A CAP of the wetland data for each of these sub-regions would thus be required to comprehend these latter correlations to within sub-region distribution, such as was performed for the Cape Flats in Section 4.3.3.

Differences in the Human Disturbance Score (HDS) between wetlands are correlated with community structure differences recorded on the Cape Flats relative to other sub-regions as evidenced by the vector length and direction in Figures 4.2 and 4.3. In both comparisons the vector direction suggests correlation of HDS with variation within rather than between sub-regions. This HDS variable perhaps illustrates the greater strength of canonical analysis than Mantel tests for finding significant correlations as the Mantel tests suggested insignificant correlation between the disturbance variables and community structure recorded in the Cape coastal lowlands (Tables 4.1 & 4.2).

4.4.3 Cape Flats wetlands

Mantel tests revealed significant correlations of the climatic variables and community structure. The inverse collinearity of evaporation with exchangeable hydrogen (H^+) and of rainfall with pH (Table 4.7) perhaps explains why the partial Mantel test suggests insignificant correlation of edaphic variables with the community structure. The canonical analysis suggested rather that individual soil variables do have strong correlations with community structure. The variation in mean annual rainfall between Strandveld and Fynbos (Table 4.8), as represented in Figure 4.4 by its collinear partner soil pH, also shows significant correlation with community structure differences between Strandveld and Fynbos associated-wetlands in the canonical analysis. Lastly, the anthropogenic variables of water quality impact and surrounding cover of woody alien vegetation also have significant correlations to community structure in the canonical analysis whilst the Mantel tests showed insignificant correlation between disturbance and community structure. These results are considered robust due to the contention of Legendre & Fortin (2010) that canonical analyses are better able to reveal significant correlations than Mantel tests.

Differences in the mean values of pH (and the collinear rainfall), potassium concentration ([K] and its collinear water soluble potassium), water soluble calcium, extent alien woody vegetation coverage in the surrounding 100 meter buffer around wetlands (and in its collinear 500m buffer) have the strongest correlations to the differences observed between the majority of the Dune Strandveld and Sand Fynbos wetlands depicted in the CAP in Figure 4.4.

Wetlands Lot04, 05 and 06 and Ken01_1, all of which are Sand-Fynbos-associated wetlands, are all closer to the Dune Strandveld wetlands than to the remainder of the Sand-Fynbos-associated wetlands in Figure 4.4. The closer proximity of wetlands Lot04, 05 and 06 to Strandveld than Fynbos wetlands, can predominantly be explained by their actual geographical proximity to the Strandveld vegetation type as well as by higher values of the spatially related pH, water soluble calcium and cation exchange capacity (CEC) than occur in the remainder of the Fynbos wetlands. Spatially related variables such as rainfall and edaphic properties thus influence the similarity of the community structure and environment of these three Sand Fynbos Lotus wetlands to be closer to the remainder of the Lotus River wetlands that are associated with Dune Strandveld.

(i) Despite the geographic proximity to the Strandveld wetlands a differential of 240mm in mean annual rainfall is apparent between the Strandveld associated wetlands (mean 553 ± 16 mm) and wetlands Lot04, 05 and 06 (mean 795 ± 15 mm), bringing these latter three wetlands closer to the Fynbos average rainfall (of 1040 ± 15 mm).

(ii) Lot04, 05 and 06 had neutral soil pH levels (mean pH 7.0 ± 0.3) differentiating them from both the acidic sand wetlands (mean pH 5 ± 0.2) associated with the terrestrial Cape Flats Sand Fynbos and from the alkaline sand wetlands (mean pH 8.2 (± 0.1) associated with Dune Strandveld.

(iii) Lot04, 05 and 06 had considerably higher levels of water soluble calcium (mean 138 ± 24 $cmol_c.kg^{-1}$) than those of the rest of the Fynbos wetlands (mean 12 ± 24 $cmol_c.kg^{-1}$); being closer to the mean value held by the Strandveld wetlands (mean 82 ± 7 $cmol_c.kg^{-1}$).

(iv) The cation exchange capacity ((CEC) as derived from the hydrogen, calcium, magnesium, potassium and sodium cations) of wetlands Lot04, 05 and 06 were, on average (5.7 $cmol_c.kg^{-1}$), closer to those of the Strandveld (3.1 $cmol_c.kg^{-1}$) than the Fynbos wetlands (2.7 $cmol_c.kg^{-1}$). The slightly greater values seen in Lot04, 05 and 06 is thought to be due to a eutrophic nutrient load in wetland Lot06 as evidenced by a phosphorous content of the sediments seven times greater than the average (9.2

mg P.kg^{-1}) recorded in oligotrophic Cape Flats wetlands. This eutrophic condition also caused a considerable increase in exchangeable potassium and whilst Lot04 and 05 had similar ambient exchangeable potassium values to the mean observed in the Strandveld sediments, Lot06 had six times this amount (0.615 cmol$_c$.kg^{-1}).

So it is apparent that Lot04, 05 and 06 have different values for these most strongly correlated environmental variables than the remaining wetlands associated with Cape Flats Sand Fynbos from either Kenilworth seven kilometres to their north and from the Dune Strandveld wetlands to which they are in much closer proximity. The gradient and spatial correlation of these environmental variables relative to these sets of wetlands serve as a good indication that the climatic and edaphic factors that are used to discriminate between the zonal units of terrestrial vegetation are at the least strongly correlated (>40%) with the distribution of wetland vegetation in the coastal lowlands of the Fynbos biome.

As for the Fynbos-associated-wetland Ken01_1, higher pH (5.7 *vs* 5) and water quality impact levels than the remainder of the Fynbos wetlands associate Ken01_1 with Lot04, 05 and 06. Similarly high percentage cover of *Typha capensis* in wetlands Ken01_1 and Lot05 and of the alien grass *Pennisetum clandestinum* in Ken01_1 and Lot06 also serves to group these four wetlands in the CAP in Figure 4.4. This group of outliers from the Fynbos wetlands serves as a good indication of the power of canonical analysis of principle coordinates (CAP) to assess correlations and differences between sampling objects.

Whilst alien woody vegetation was only present around the wetlands from the Kenilworth Race Course (representing most of the Sand Fynbos wetlands) it had recently been cleared from the wetlands of the Lotus River and Kuils River Floodplains (that represent the Strandveld wetlands). Phosphorus content of the litter layer (1050µg.kg^{-1}) beneath an infestation of alien acacias (*A. cyclops*) was shown to be greater than that of litter under indigenous Fynbos vegetation (360µg.kg^{-1}) (Witkowski & Mitchell 1987). The recent removal of the alien acacias from around many of the Strandveld wetlands will not have removed the litter which may yet be impacting on the phosphorus content of the litter, sediment and water. Hence the significant correlation of the cover of alien vegetation around impaired Fynbos wetlands with the cover of species within them is considered an important relationship differentiating impaired from minimally impaired wetlands. This impact is considered as likely to be important for both the Strandveld and Fynbos associated wetlands, despite only being testable for Fynbos wetlands in the current study.

4.5 Conclusions

Differences caused by climate and geology correlate with phytogeographical discontinuities within the Cape Lowland Freshwater vegetation. Spatially related environmental variables correlate with differences in the floristic communities of wetlands from different spatial units, namely the sub-regions, bioregions and edaphic substrates within the Cape coastal lowlands. At the biome scale increasing evaporation, from the Agulhas Plains in the south-east to the Verlorevlei in the north-west of the Cape coastal lowlands, has a significant correlation with observed wetland plant community structure and species beta diversity variation between sub-regions. This correlation is suggestive of the potential of some climatic responsibility for the observed beta diversity or turnover and resultant floristic differences between these areas (Chapter 3). In the Fynbos biome phytogeographical units grouped by climate and geology, such as the Bioregions (Rutherford *et al.* 2006) or the NFEPA wetland-vegetation-groups (Roux *et al*. 2006), contain a considerable range of natural climatic and edaphic variability. Such a range could mask changes induced by anthropogenic disturbance and thus reduce the potential to identify vegetation attributes that correlate with anthropogenic disturbances that could be used for phytoassessment. At the regional scale, within the Cape Flats sub-region, climatic (rainfall) and edaphic (pH, $Ca_{\text{water soluble}}$, [K]) differences between the Dune Strandveld and Sand Fynbos wetlands (as representative of West Strandveld and South West Fynbos bioregions) are correlated with the observed variations in community structure of each spatially distinct vegetation unit. These discontinuities in units of wetland vegetation are aligned or correspond with discontinuities or differences between the (intra)zonal terrestrial units of vegetation. Phytogeographical units of wetland vegetation associated with the (intra)zonal terrestrial vegetation units in the Cape coastal lowlands have limited natural edaphic and climatic variability. Within these units of wetland vegetation anthropogenic impacts (disturbances) were shown to have significant correlations with differences between impaired and unimpaired vegetation (Figure 4.4). Spatial subsets based on association with (intra)zonal terrestrial vegetation units, would therefore be useful for the determination of phytoassessment metrics. At the local landscape scale, within wetlands associated with these terrestrial vegetation units in the Cape Flats, water quality and the percentage cover of alien woody vegetation around wetlands are human disturbances that correlated with floristic differences between reference and impaired wetlands. The number of wetland replicates sampled in the Cape Flats (Table 2.2) and the differences within this subset between reference and impaired wetlands (Figures 3.20 and 4.4) suggests the potential to identify species attributes characteristically associated with each of these environmental conditions. This possibility is explored in Chapter 5.

CHAPTER 5

TOWARDS THE USE OF PLANTS IN THE ASSESSMENT OF WETLAND CONDITION

Measurement of the difference in patterns of plant communities in reference relative to disturbed wetland habitat of the same type has been used to assess wetland ecosystem condition in the process typically known as bioassessment (e.g. Mack 2000, Simon *et al.* 2001, Gernes & Helgen 2002). In these studies, plant based bioassessment, or phytoassessment, was successfully developed in phytogeographical regions with limited natural variability. In this chapter I attempt to identify species, life-history groups (*sensu* Galatowitsch 2000), or other vegetation attributes that are characteristically associated with impaired relative to minimally impaired wetlands. The wetlands associated with the Cape Flats Dune Strandveld and Sand Fynbos terrestrial vegetation units were both shown (Chapter 3) to hold relatively homogenous sets of wetland vegetation that were significantly different from one another and that showed significant differences between reference and impaired wetlands. The distribution of wetland vegetation in the Dune Strandveld and Sand Fynbos were also shown (Chapter 4) to correlate with different environmental variables. Whilst all analyses to this point have used the community structure of the whole wetland or of entire hydrological habitats of a wetland this chapter focuses at a finer scale, basing comparisons of the community structure on the individual relevés made in each characteristic stand of vegetation. This finer scale provides greater detail within which to search for community differences between hydrological zones and between impaired and reference conditions. This chapter explores these data sets with the objective of examining their potential to provide measurements of vegetation attributes (metrics) that would facilitate phytoassessment of environmental condition in wetlands of the Cape Flats.

5.1 Methods for determining comparable wetland vegetation units

5.1.1 Hydrological differences

As mentioned in Chapter 1 (Section 1.5.3), comparison of the whole complement of vegetation of different wetlands only makes sense when the hydrological habitats present in each wetland are the same or when the vegetation of each of these habitats are not

significantly different. The differences in vegetation sampled in supralittoral and littoral habitats were significant but not considerable when compared across the whole of the Cape coastal lowlands using ANOSIM (Section 3.3.2), yet I hypothesized that considerable differences would be apparent between these hydrological habitats within phytogeographical units of vegetation that were naturally homogenous. To test this hypothesis within the Cape Flats data set, vegetation samples (relevés) were examined for community differences between hydrological habitats from wetlands associated with Dune Strandveld and Sand Fynbos and from wetlands at the interface of these two units, from sites that had either reference or impaired environmental conditions. The spatial hierarchy of these relevés for the Cape Flats data set is presented in Table 5.1.

Table 5.1: Spatial hierarchy and number of vegetation samples associated with different environmental-disturbance conditions in different hydrological habitats and with different units of terrestrial vegetation from the central or Cape Flats sub-region of the Cape coastal lowlands.

Sub-region	Associated Terrestrial Vegetation Unit	No. of samples and Habitat	No. of samples and Condition
Cape Flats	Cape Flats Dune Strandveld	52 Supralittoral	26 Reference
			26 Impaired
		54 Littoral	26 Reference
			28 Impaired
	Cape Flats Sand Fynbos	50 Supralittoral	18 Reference
			32 Impaired
		40 Littoral	8 Reference
			32 Impaired
	Cape Flats Strandveld and Fynbos interface	19 Supralittoral	8 Reference
			11 Impaired
		17 Littoral	4 Reference
			13 Impaired

The differences in community structure between these numerous units of vegetation can be ascertained with the use of a multivariate analysis procedure such as PERMANOVA (Anderson *et al.* 2008). Variation caused by geographical distance was dealt with by using spatial variables of longitude and latitude as co-variables before searching for variation created by other differences (species or environmental) thereby reducing the effects of spatial autocorrelation among samples (Legendre & Fortin 1989, Legendre *et al.* 1990). The effects of spatial autocorrelation among samples from different vegetation units were also addressed in the generation of significance in the PERMANOVA design by nesting the habitat units within a given associated terrestrial unit of vegetation (Anderson *et al.* 2008). For nested subsets, the samples of each subset are only permutated amongst themselves in the generation of significance levels, thereby accommodating for the lack of spatial independence of the subsets (Cliff & Ord 1981, Legendre 1993).

5.1.2 Community Structure differences

Permutational Analysis of Variance is able to pick differences in community structure between sampled objects that are not apparent with unconstrained ordination (Anderson *et al*. 2008). Ordination of the samples of any given unit of vegetation was performed to reveal whether the magnitude of separation, initially made apparent by PERMANOVA, between the reference and impaired vegetation samples is considerable enough such that these differences would be apparent during a field sampling process such as performed for phytoassessment. If differences in community structure between reference and impaired samples are pronounced enough such that separation between them is apparent with unconstrained ordination then the identification of species that are significantly correlated with community differences between groups is likely to be easier. A difference of two or more steps on the Braun Blanquet cover / abundance scale (Table 2.8) (e.g. 2m (<5%) to 2b (12.5 – 25%)) is considered large enough difference to distinguish significant difference for metric species (Mack 2007, after Westhoff & van der Maarel 1978).

Examination of the difference between community structure using life-history groups rather than species may reduce the differences that exist between any of these units of vegetation, however, such an approach that lumps species is also likely to reduce the potential to ascertain the impacts of natural *vs* anthropogenically driven variation. The life-history groups are however useful in determining species with like response to anthropogenic disturbance and are thus employed in a search for attributes with discriminatory potential between reference and impaired conditions as described in Section 5.1.4.2.

5.1.3 Drivers of difference

Within ordinations separate groups of samples from the same category of environmental condition (or anthropogenic disturbance) may indicate natural causes of difference which may or may not be explainable by the environmental variables measured at each site. In my opinion therefore the use of differences in floristic community structure (or diversity) between reference and anthropogenically impaired communities to assess environmental condition can only be justified once significant correlation with anthropogenic stressors has been proven. Whether natural or anthropogenic influence is responsible for differences in floristic community structure between reference and impaired relevés in

any otherwise homogenous unit of vegetation was explored in this chapter with Canonical Analysis of Principal Coordinates (CAP) (Anderson & Willis 2003).

5.1.4 Characteristic vegetation attributes

If homogenous units of vegetation hold different floristic communities within reference and impaired samples then for purposes of phytoassessment, attributes of the vegetation that represent these differences can be used as metrics or measurements of condition. A species that is more often associated with reference than impaired conditions can be said to be discriminatory between these two conditions and is therefore considered to represent an indicator species (Karr 1987) or bioindicators (Karr & Chu 1999). Similarly any attribute of the vegetation that is discriminatory between two disturbance groups is useful as an indicator and a metric for measuring or scoring condition.

Diversity indices, species counts and cover values using the relevé data were used in the identification of discriminatory vegetation attributes. Thereafter significant differences between disturbance categories were established with pair-wise t-tests using the relevés as permutatable replicates for each category. The use of t-tests, with significance values generated using permutation, removes the need to "jackknife" relevé values (of diversity or species cover) before determining significant differences. These t-tests are therefore relatively robust even at limited sample sizes ($10<n<100$) (Hope 1968, Manly 1997).

5.1.4.1 Species attributes

Species that are discriminatory between reference and impaired conditions were identified using similarity percentage analysis (SIMPER) in PRIMER-E (Clarke & Warwick 2001). The SIMPER analysis provides a list of species that are characteristic of a group or discriminatory between two groups. While identification of characteristic species could also be achieved by Braun-Blanquet table arrangement or with divisive classification as performed by Two Way Indicator Species Analysis (TWINSPAN) (Leps & Smilaur 2003), I consider the identification of discriminatory species to be of greater use for development of phytoassessment metrics. SIMPER is able to ascertain characteristic species of a group or species that are discriminatory between two groups. Species that have consistently high cover/abundance throughout the samples of a group (i.e. reference or impaired) are considered characteristic of the group (Clarke & Warwick 2001). A consistent high cover/abundance means that such a species will have a low standard deviation in contributing to group similarity and therefore the ratio of the species contribution to group average similarity (group average squared distance) to its standard

deviation of group similarity (Sim/SD) will be high. The high Sim/SD of a characteristic species does not guarantee the species is a good discriminator between groups as it may occur with equal cover/abundance in both groups and therefore not distinguish between them (Clarke 1993). It is also possible, however, by the same procedure to determine the species that are discriminatory between different groups from the ratio (Diss/SD) of the contribution of a species to group average dissimilarity (Diss) relative to the standard deviation of its contribution to group average dissimilarity (SD) (Clarke & Warwick 2001).

Species that contribute much to the difference between the groups will be of low discriminatory use when they do not occur consistently enough (i.e. they do not have high enough fidelity) across the samples of either group (such species would have high standard deviation and thus low Diss/SD). Therefore the higher the Diss/SD ratio the more reliable the species is as a discriminator.

Whilst the SIMPER analysis reveals the "average" cover value of a species within a group (e.g. the average percentage cover of *Typha capensis* within all impaired littoral relevés from Kenilworth), this value may under-represent the cover in the actual relevés in which a species was recorded as each species is not recorded in each relevé. This problem is not encountered with TWINSPAN as the averaging of values from numerous relevés to represent a group is not performed. For any species not present in all relevés, the average value per disturbance category as determined with SIMPER is deflated to a level considerably below that which was typically encountered in relevés where one of these species occurred. Hence the discriminatory species chosen by the SIMPER analysis based on average cover were not in all cases practical for discriminating between reference and impaired relevés or wetlands. In order to improve resolution per relevé the "typical" cover-value was generated based on the average cover in all relevés within which a species was encountered. Typical and average cover values were used to generate a graph of the species most discriminatory between reference and impaired samples. These "average" and "typical" cover values for discriminatory species were then used to identify possible metrics of environmental condition. The use of either average or typical measures per relevé (see Section 5.2.4) or per wetland each require a different approach in the way a metric is used for phytoassessment purposes.

5.1.4.2 Diversity attributes

An exploration of the differences in species diversity among reference and impaired relevés has the potential to reveal metrics for use in the development of an index of environmental condition. To this end an analysis of the various measures of species diversity was performed using DIVERSE in PRIMER-E (Clarke & Warwick 2001) and some basic interrogation of the data set. The following diversity measures were determined per relevé:

- Total number of species (S);
- Species richness as determined by Margalef's index (d) (Margalef 1975);
- Species diversity: Shannon-Weiner (H′) (Shannon & Weaver 1949) and/or Simpson's diversity index (Dominance (λ) or Evenness (1- λ)) (Simpson 1949).

These diversity indices were determined for the entire set of species and for species subsets based on the life-history groups of Galatowitsch *et al*. (2000) as based on growth forms (i.e. graminoids or shrubs), structural aspects of plant form (woodiness), life-history attributes (annuals *vs* perennials), affinity to the wetland habitat (obligate or facultative affiliation to wetland habitat (*sensu* Reed 1988)), and origin (indigenous *vs* alien).

The area that a species covers, or the number of individuals of a given species, can be used interchangeably in the generation of diversity indices (Whittaker 1965). Although the Braun-Blanquet cover scale provides more consistently comparable data than absolute measures of cover when used by multiple samplers for phytoassessments (USEPA 2002b, Mack 2007) it has been reported (Magurran 1992) as producing a biased result if used (in place of abundance) in conjunction with diversity indices as the scale is not linearly correlated with abundance. Braun Blanquet cover values have, however, been successfully used in developing diversity index based ecological indicators for aquatic ecosystems (Madden *et al*. 2009). In the present study, in which diversity measures are to be compared between samples in order to determine whether they can be used as a metric of difference between disturbed and undisturbed vegetation communities, the bias is considered likely to impact all samples equally and is thus not considered important. A brief description of the diversity measures used in the present research are provided below.

i. Species richness including Margalef's index

Richness is a measure of the number of different kinds of organisms present in a particular area. Species richness for a specific group of organisms is the number of

different species of that type of organism present. Whilst species richness is often given simply as the number of taxa, this measure is obviously very dependent on sample size and therefore also on sampling effort. Margalef's index (d) determines species richness relative to the total median cover in a sample (N), resulting in a proportion based index that is thus less dependent on sample size and effort. Margalef's index is based on the number of different species within a sample standardized against the total cover each species occupies within that sample; therefore, species with little cover add proportionately less to the index. Per sample, Margalef's richness is calculated by dividing the total number of species less one, by the natural logarithm of their total cover or abundance: $\mathbf{d = (S-1)/Log_eN}$

ii. Species Diversity

Biological diversity can be quantified in many different ways. The two main aspects taken into account when measuring diversity are richness (as described above) and evenness. Evenness compares the abundance of each species present with that of each other species. If all species have similar cover/abundance in a specific area then the evenness is higher. If there is one species that is highly dominant, it will reduce the evenness, and in many cases also the richness, since the dominant species will out-compete many other species including similar types of organisms. The relative cover or relative abundance of the different species making up the richness of the wetlands is thus a measurement of evenness (*sensu* Simpson 1949). Diversity indices incorporating both evenness and richness used in the search for phytoassessment metrics were those developed by (a) Shannon-Wiener and (b) Simpson.

a.) Shannon-Wiener diversity index

The Shannon-Wiener index (H′) measures overall biodiversity. H′ is calculated per sample (wetland) as the sum of the proportion of the total cover/abundance arising from the i^{th} species (p_i) multiplied by the natural logarithm of this proportion: $\mathbf{H' = -\sum_i p_i \log_e(p_i)}$. H′ is maximized when all species have the same number of individuals. For example, it is biggest when a wetland has 4 aquatic herbs, 4 graminoids, and 4 shrubs. H′ is smaller when a wetland has 1 aquatic herb, 2 graminoids, and 5 shrubs despite the fact that both wetlands have 8 inhabitants. The Shannon-Wiener index is, however, sensitive to the degree of sampling effort (Clarke & Warwick 2001).

b.) Simpson's Diversity Index - Dominance or Evenness

Simpson's diversity index measures the richness and percentage cover of species in a sample or habitat. The index assumes that the proportional cover/abundance of species

in an area indicates their importance to diversity within that ecosystem. Due to the importance of proportions, unlike the previous Shannon-Wiener index, Simpson's measure of species diversity is less sensitive to the degree of sampling effort (Clarke & Warwick 2001). Simpson's diversity is a dominance index, in the sense that its largest values correspond to assemblages whose total cover/abundance is dominated by one, or very few, of the species present. Simpson's dominance as determined per sample (i.e. per wetland or per relevé) is the sum of the square of proportion of the total cover/abundance arising from the i^{th} species (pi): **Dominance = $\lambda = \sum p_i^2$**

The range of this function is between 0 and 1, with more biologically diverse samples scoring near 0 and more monotypic scoring near 1 where 1 represents no diversity or that only one species is present (Clarke & Warwick 2001). The inverse of this latter index is a measure of *evenness*, with largest value when all species have the same cover/abundance: **Evenness = $1 - \lambda = 1-(\sum p_i^2)$**

5.2 Results

5.2.1 Hydrological differences

A PERMANOVA of the differences between vegetation units separated by association with different anthropogenic disturbance conditions, different hydrological habitats and with different edaphic substrates as associated with the terrestrial vegetation units is presented in Table 5.2. Data generated by PERMANOVA based on the Cape Flats vegetation samples from different units of vegetation (see Table 5.1) indicate:

#1 wetland vegetation associated with each of the Dune Strandveld, Sand Fynbos and the samples at the interface of these vegetation units represent significantly different floristic communities (pseudo-F = 2.3, $p < 0.03$);

#2 across the collective of all of these vegetation relevés, the impaired and reference samples are not significantly different communities;

#3 each hydrological habitat (as nested within each associated terrestrial vegetation unit) holds significantly different communities (pseudo-$F_{1, 2} = 4.6$, $p = 0.001$);

#4 within the combined samples of both supralittoral and littoral hydrological habitats from within each associated terrestrial vegetation unit no significant difference in communities was apparent between reference *vs* impaired samples; however,

#5 within each hydrological habitat as isolated subsets of each terrestrial vegetation unit [habitat (associated terrestrial vegetation unit) x condition], differences in vegetation relevés between reference and impaired conditions are apparent and significant (pseudo-$F_{1, 2} = 1.8$, $p < 0.001$).

Table 5.2: PERMANOVA of the vegetation sample data for the Cape Flats. Relationships that are significant at p < 0.001 are marked with **, relationships that are significant at p < 0.05 are marked *.

Species cover/abundance		Degrees of freedom	Pseudo-F	p-value
#1	Associated-Terrestrial Vegetation Unit	2	2.3	0.03*
#2	Condition	1	1.8	0.15
#3	Habitat(A-T Veg Unit) †	3	4.6	0.001**
#4	A-T Veg Unit x Condition	2	1.7	0.14
#5	Habitat(AT Veg Unit) x Condition	3	1.8	0.001**
	Residual	218	-	-
	Total	231	-	-

† A-T Veg Unit = wetland samples from within an Associated-Terrestrial Vegetation Unit

These results reiterate the findings reported in chapter 3, namely that the vegetation associated with each terrestrial vegetation unit is different; and that when the whole data set is viewed collectively no difference is apparent between reference and impaired samples. These analyses also show that the communities associated with each hydrological habitat are different and that when vegetation samples from supralittoral and littoral habitats associated with each terrestrial vegetation unit are examined independently, significant difference in community structure is apparent between the reference and impaired samples. These differences were restricted to only a portion of the vegetation units listed in Table 5.1 and *a posteriori* pair-wise analysis (Table 5.3) revealed the significance level of the differences between reference and impaired samples within each of the habitats associated with each of the terrestrial vegetation units. The wetlands associated with Cape Flats Dune Strandveld have significant community structure differences between samples from reference and impaired conditions in both their supralittoral and littoral habitats. The vegetation samples from the supralittoral wetland habitat associated with Cape Flats Sand Fynbos also show considerable differences between reference and impaired conditions. Any of these three sets of samples with significant differences in community structure between reference and impaired conditions represent vegetation units within which to search for vegetation attributes with discriminatory association for these conditions.

Table 5.3: Comparison of the vegetation associated with different terrestrial vegetation units, hydrological habitats and environmental conditions using pairwise t-tests in PERMANOVA. Vegetation units that are significantly different at p < 0.001 are marked with **, relationships that are significant at p < 0.01 are marked *.

Associated-Terrestrial Vegetation Unit	Habitat	Condition Groups	t-test	p-value
Cape Flats Dune Strandveld	Supralittoral	Reference *vs* Impaired	1.5	0.01*
	Littoral	Reference *vs* Impaired	1.7	0.002*
Cape Flats Sand Fynbos	Supralittoral	Reference *vs* Impaired	1.9	0.001**
	Littoral	Reference *vs* Impaired	1.5	0.054
Interface Fynbos & Strandveld	Supralittoral	Reference *vs* Impaired	1.0	0.5
	Littoral	Reference *vs* Impaired	1.2	0.2

5.2.2 Ordination of community structure differences

i. ***Littoral Cape Flats Dune Strandveld***

Ordination of the littoral samples of the Cape Flats Dune Strandveld shows considerable intermingling of samples from reference and impaired conditions (Figure 5.1). This suggests that whatever differences exist in the community structure of reference relative to impaired samples, it is limited and does not create easily differentiable stands of vegetation. Only samples from littoral habitat with maximum potential annual inundation depth of ≤400mm were used for this series of analyses because the maximum depth of inundation of the reference relevés was ≤400mm.

Differences between relevés of littoral Dune Strandveld vegetation from different localities on the Cape Flats are apparent in the ordination in Figure 5.1; emphasizing the importance of spatially related environmental gradients (Chapter 4). The relevés from the spatially disparate Lotus River and Kuils River (Driftsands) floodplains are somewhat intermingled. A significant difference exists between the reference samples from Lotus River and Driftsands as determined with ANOSIM (R=0.17, p<0.05) and an even greater difference exists between the reference Lotus and impaired Driftsands samples (R=0.25, p<0.05). Whilst impaired littoral relevés were assessed at the Lotus River, these relevés were all at the interface between Strandveld and Fynbos and are distinct from either unit. No other impaired Lotus relevés were assessed and therefore, for the purposes of phytoassessment development, these impaired littoral relevés at the interface of Strandveld and Fynbos cannot be compared with littoral vegetation within Dune Strandveld (Chapter 3.2.3.3). The intermingling of reference and impaired Driftsands Strandveld-associated littoral relevés indicates that it would be difficult to establish reliable metrics for phytoassessment from this data set.

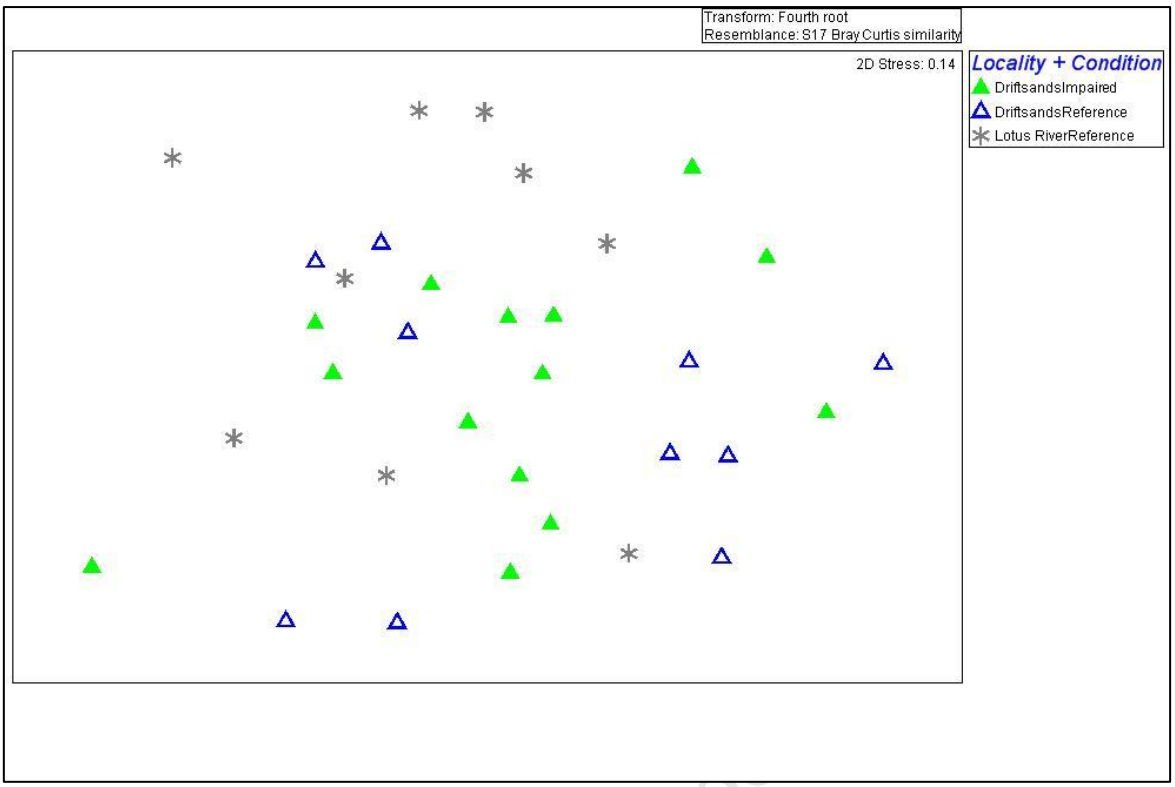


Figure 5.1: Non-metric multi-dimensional scaling of littoral samples with reference and impaired environmental conditions associated with the Cape Flats Dune Strandveld as sampled in the Lotus River floodplain and at Driftsands on the Kuils River floodplain.

ii. ***Supralittoral Cape Flats Dune Strandveld***

Similar intermingling is evident in the impaired and reference supralittoral samples of the Dune Strandveld of the Cape Flats, suggestive of either limited impact or of an advanced stage of deterioration of all wetlands from a truly reference or entirely un-impacted condition. The ordination of this set is not shown for the sake of brevity.

iii. ***Supralittoral Cape Flats Sand Fynbos***

Ordination of the supralittoral vegetation samples associated with Cape Flats Sand Fynbos shows almost complete separation between reference and impaired relevés that is suggestive of considerable differences in community structure (Figure 5.2). It is again evident that considerable community differences exist between samples from the Fynbos sites sampled at Kenilworth and those sampled at Lotus River, some seven kilometres further south. No reference relevés within Fynbos were sampled at the Lotus River floodplain as at the time of sampling Mucina *et al.* (2006a) considered wetland taxa as not necessarily Fynbos, Strandveld or Renosterveld affiliates and all wetlands of a given HGM type within a given locale were considered likely to hold similar vegetation.

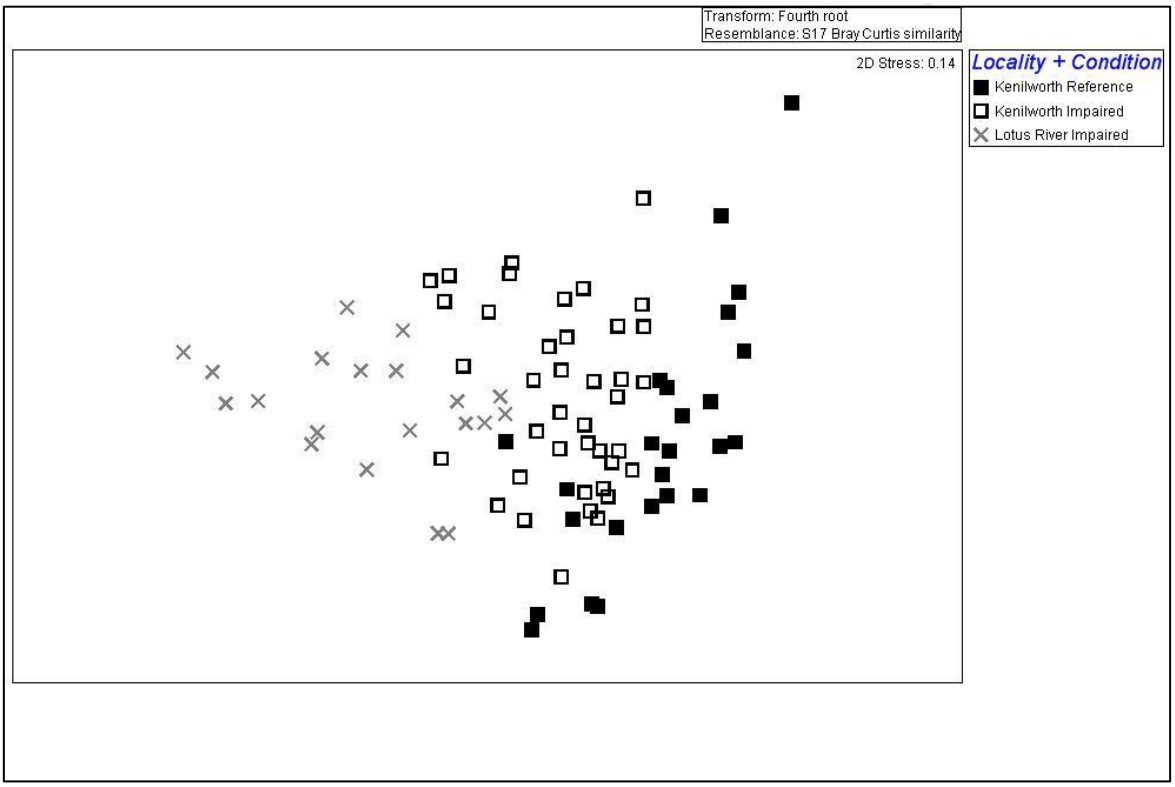


Figure 5.2: Non-metric multi-dimensional scaling of supralittoral samples with reference and impaired environmental conditions associated with the Cape Flats Sand Fynbos as sampled in Kenilworth and Lotus River.

The impaired Fynbos-associated samples from Lotus River and Kenilworth were considerably different as shown with ANOSIM (R=0.33, p<0.001). Even greater differences were evident between the impaired Lotus and the reference Kenilworth samples (R=0.64, p<0.001) as determined with ANOSIM. Only explanation of the community difference between Kenilworth reference and impaired samples was thus attempted as the Lotus samples add natural variation that may mask the influence of anthropogenic disturbance. The separation between Kenilworth impaired and reference

relevés evident in Figure 5.2 suggested that good potential exists for identifying vegetation attributes that are discriminatory between impaired and reference samples for the supralittoral Kenilworth Sand Fynbos data set.

5.2.3 Discriminatory environmental differences

The supralittoral relevés from Kenilworth are predominantly from the flat edge of sandy Depressions but include some relevés from sandy Flats (*sensu* the HGM units of SANBI 2009). The vegetation is mostly dominated by Cape Lowland Freshwater vegetation, but Vernal Pool and terrestrial vegetation (with many typical terrestrial *Capensis* elements) dominate in relevés from the Flat HGM unit. The above ordination of the supralittoral Kenilworth relevés (Figure 5.2) showed relatively distinct separation of reference (n=18) and impaired (n=22) relevés. A fern-dominated sample Ken10_14 was an outlier in the species ordination (even further removed than Ken10_13 that is at the top right of Figure 5.2) and was thus excluded from this ordination and further analyses.

The flat and vernal pool samples had no surface water and water environmental variables were therefore not measured at these two wetlands. Environmental variables with strong correlations to community differences between reference and impaired relevés were thus sought using two separate data sets: (i) a data set of anthropogenic, climatic and edaphic variables (in the "edaphic" data set) and (ii) another analysis with a reduced number of samples using water variables instead of the edaphic variables (in the "water" data set). The edaphic, climatic and disturbance variables in these analyses are those listed at the start of Section 4.2 and listed in Sections 2.2.6.1, 2.2.7(i) and 2.2.7(iii). The water variables consist of those listed in Table 2.6 of section 2.2.5. A number of variables were collinear with other variables as presented in the triangular matrices in Tables 5.4 and 5.5 for the edaphic and the water data sets respectively. Sand, water soluble sodium and magnesium and rainfall were removed from the "edaphic" environmental matrix to reduce collinearity, whilst dissolved oxygen and rainfall were removed from the "water" data set. Variables removed from the data are thereafter represented in analyses by their collinear partner variables as shown in Tables 5.4 and 5.5.

***Table 5.4*: Collinear environmental parameters including edaphic variables in the supralittoral Cape Flats Sand Fynbos associated relevés at Kenilworth. Bold and Underlined figures show positive Pearson correlations greater than 90% or negative (inverse) correlations in excess of – 90%.**

	% Silt	% Sand	Resistance	Exca Na	Na water soluble	Mg water soluble	Evap'n	Rainf'l
%_Silt	-							
%_Sand	**-0.94**							
Resistance	-0.46	0.54						
Exca Na	0.69	-0.74	-0.83					
Na water soluble	0.51	-0.63	**-0.93**	**0.91**				
Mg water soluble	0.49	-0.55	**-0.90**	0.78	0.84			
Evaporation	-0.10	0.04	-0.14	-0.02	0.04	0.12		
Rainfall	0.10	-0.03	0.15	0.01	-0.07	-0.13	**-0.997**	-

Exca = exchangeable cations

***Table 5.5:* Collinear environmental parameters in the environmental data set including water variables for the supralittoral Cape Flats Sand Fynbos associated relevés from Kenilworth. Bold and Underlined figures show negative (inverse) Pearson correlations of – 90% or more.**

	Dissolved Oxygen	Physical Disturbance	Evaporation	Rainfall
Dissolved Oxygen	-			
Physical Disturbance	**-0.96**			
Evaporation	-0.04	0.16		
Rainfall	-0.01	-0.11	**-0.997**	-

i.) Discriminatory edaphic, climatic and anthropogenic variables

A plot of the CAP separation of the different supralittoral relevés is presented below (Figure 5.3) with vector overlays of the sphericised environmental variables that best correlated to the sphericised community data (species cover) that are represented in the ordination of relevés (canonical correlation > 0.35). The cut-off of 35% correlation was arbitrarily chosen and decreasing this value increases the number of environmental vectors placed in the graph, each one of which will correlate with the sphericised species ordination. The anthropogenic variables of buffer width (loss in extent thereof), physical disturbance and water quality impacts have the greatest correlations to the differences between reference and impaired relevés. The vector direction for the concentration of potassium [K_{Loge}] (or log(K+0.1) in Figure 5.3) suggests correlation with intra- rather than inter-category community variation.

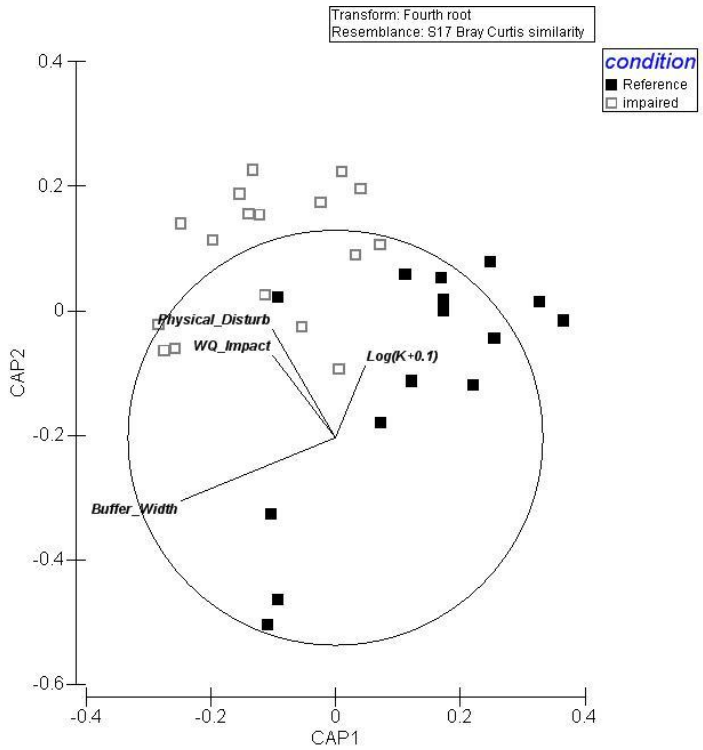


Figure 5.3: Canonical Analysis of Principal Coordinates of the environmental and species data of the supralittoral Fynbos samples from Kenilworth. Vectors of environmental variables with ≥ 35% correlation with the sphericized species resemblances overlay an ordination of these relevés.

A summary of the mean value within reference and impaired relevés for the four environmental variables of the "edaphic" set with canonical correlations of greater than 0.35 (as shown in Figure 5.3) are provided in Table 5.6. Potassium concentration was the only edaphic variable that had a correlation of ≥35% with the pattern of community structure in these supralittoral Fynbos samples and the remainder of these variables are measures of anthropogenic disturbances. Potassium concentration may also relate to anthropogenic influence or disturbance as was the case in Lot06 (Section 4.3.3).

Table 5.6: Mean values (± Standard Error) for environmental variables with strong correlation with the differences between the reference and impaired supralittoral wetland relevés from Kenilworth Sand Fynbos using the "edaphic" data set.

Variable	**Physical Disturbance**	**Water Quality**	**Buffer extent (loss thereof)**	**Potassium**
Unit	(Score)	(Score)	(Score)	(mg K. kg^{-1})
Reference	18 (1)	22 (0.5)	4 (1)	13.8 (3.3)
Impaired	43 (2)	48 (2)	10 (0.8)	14.3 (2.3)

ii.) Discriminatory water, climatic and anthropogenic variables

Using the relevés for which water physico-chemistry data were recorded CAP indicated that anthropogenic impacts had the greatest correlations with the resemblances of community structure in reference *vs* impaired relevés as displayed in Figure 5.4. Canonical correlations of the vector overlays of the sphericised environmental *vs* sphericised species variables (cc > 0.5) are displayed in Figure 5.4.

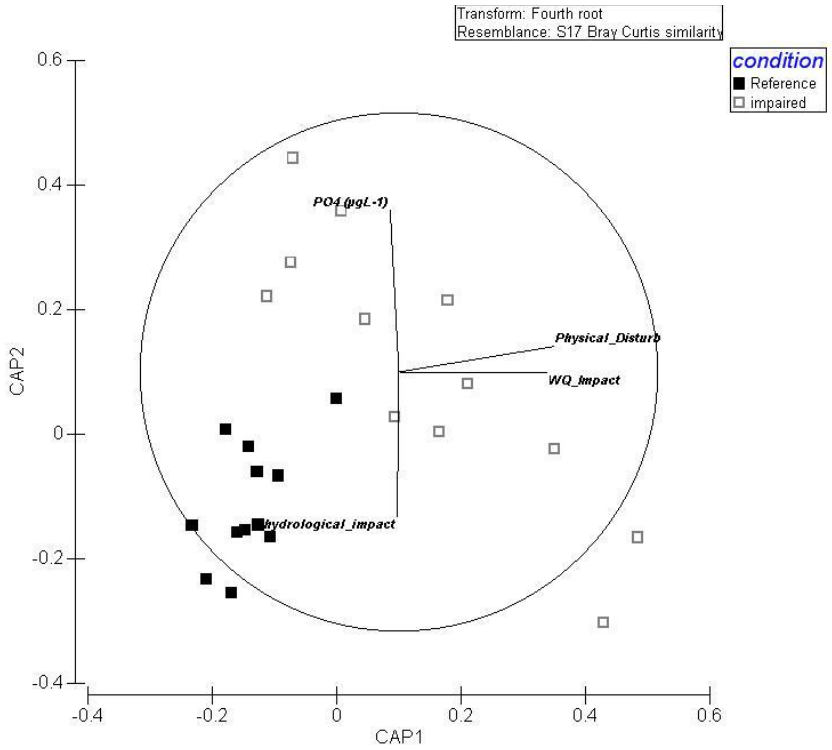


Figure 5.4: Canonical correlation ordination plot relating reference and impaired relevés to influential water, climatic and anthropogenic environmental variables for the supralittoral-Kenilworth relevés. Vector overlays are canonical correlations (*cc>0.5).*

A summary of the mean value within reference and impaired relevés for the four environmental variables from the "water" variable set with canonical correlations of ≥ 50% (a large effect size *sensu* Cohen (1988)) (as shown in Figure 5.4) are provided in Table 5.7.

Table 5.7: Mean values (± Standard Error) for environmental variables with strong correlation with the differences between the reference and impaired supralittoral wetland relevés from Kenilworth Sand Fynbos using the "water" data set.

Variable units	**Hydrological Impact** (Score)	**Water Quality** (Score)	**Physical Disturbance** (Score)	**Phosphates (PO_4)** (µg PO_4. L^{-1})
Reference	- 11 (2)	19 (3)	23 (1)	1.9 (0.3)
Impaired	- 26 (3)	44 (2)	50 (3)	14.3 (3)

5.2.4 Characteristic vegetation attributes

i.) Discriminatory species in Kenilworth supralittoral relevés

Species with discriminatory cover between reference and impaired Sand Fynbos associated supralittoral relevés at Kenilworth were sought using SIMPER. The SIMPER analysis revealed 16 species with discriminatory potential between the reference (n=17) and impaired (n=22) Kenilworth supralittoral relevés as shown in Figure 5.5. As mentioned in Section 3.3, all alien species are marked with an asterisk in all figures and text where they are referred to by name. The average and typical cover of these species in each disturbance category is represented in Figure 5.5 with the length of the bar being equivalent to the magnitude of cover.

As is apparent from Figure 5.5, the mega-graminoid *Pennisetum macrourum*, the graminoids *Pentaschistis pallida, Juncus capensis* and *Restio quinquefarius*, the shrubs *Berzelia abrotanoides*, *Psoralea pinnata* and *Rhus laevigata* var. *laevigata*, the herb *Plecostachys serpilifolia*, the fern *Histiopteris incisa* and the sedge *Chrysitrix capensis* all occurred with greater average and typical abundance in the reference than in the impaired relevés.

The mega-graminoid *Typha capensis*, the indigenous lawn grass *Cynodon dactylon*, the alien tussock grass *Phalaris aquatica**, the sedges *Cyperus sphaerospermus* and *Cyperus textilis*, and the arum lily *Zantedeschia aethiopica* all occurred with greater average and typical cover in the impaired than in the reference relevés.

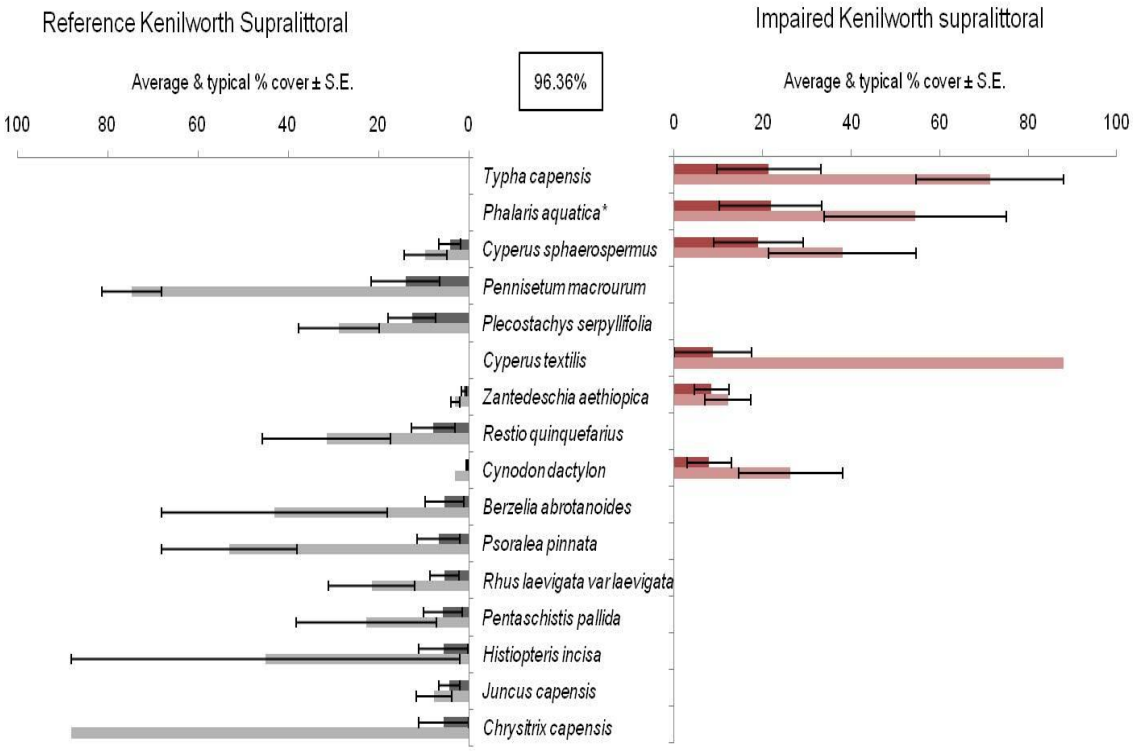


Figure 5.5: Average (dark bars) and typical (light bars) species cover (± standard error) in the reference and impaired supralittoral relevés of the Sand Fynbos associated wetland vegetation at Kenilworth on the Cape Flats. The dissimilarity percentage between the species assemblages of the different disturbance categories is presented in the rectangle at the top of the graph.

Species without error bars have standard error of zero, often meaning they were present only in a single relevé. *Chrysitrix capensis* and *Cyperus textilis* were present only in single relevés and have, therefore, potentially low fidelity to reference and impaired environmental conditions respectively. They do, however, represent typical stands of supralittoral vegetation at Kenilworth. Consideration of the validity of the use of such data is discussed in Section 6.6.7. The clear separation between reference and impaired relevés in the nMDS ordination (Fig 5.2) and CAP ordinations (Fig 5.3 and 5.4) suggest that the species displayed in Figure 5.5 are relatively good discriminators between reference and impaired environmental conditions in the Sand Fynbos associated supralittoral wetland habitat at Kenilworth.

ii.) Diversity differences between reference and impaired relevés

In total 112 species were recorded in the supralittoral relevés associated with Sand Fynbos at Kenilworth. Measures of diversity of various groups of taxa that occurred with significantly different value in reference and impaired supralittoral-Kenilworth habitat are reported in Table 5.8.

Table 5.8: Diversity differences between reference and impaired Sand-Fynbos-associated supralittoral relevés at Kenilworth. Values in reference and impaired categories represent the average per relevé (±S.E.). Whilst the cover of woody taxa was not shown to be significantly different using a posteriori PERMANOVA, it was significant when assessed using ANOSIM and the values for this ANOSIM are placed in brackets.

Diversity variable	Taxa groups	Reference	Impaired	t-test	p-value
cover	Annuals	2.4 (0.92)	19 (6.4)	2.2	0.03
number	All aliens	0.4 (0.2)	2 (0.4)	3.5	0.001
cover	All aliens	0.7 (0.3)	21 (6)	2.8	0.01
Diversity**	All aliens	0.04 (0.04)	0.4 (0.1)	2.4	0.02
cover	Woody taxa	18(6)	5(4)	(R = 0.2) 1.7	(p < 0.01) 0.1
cover	Sclerophyllous shrubs	13 (6)	0.6 (0.4)	2.5	0.001
number	Leafless graminoids	0.7 (0.2)	0.1 (0.06)	2.7	0.01
cover	Leafless graminoids	14 (6)	1 (0)	2.3	0.02
number	Alien graminoids	0.1 (0.1)	0.9 (0.2)	3.5	0.002
cover	Alien graminoids	0.3 (0.2)	15 (6)	3.5	0.002

**Shannon Wiener diversity ($\log_e$)

- Greater mean cover of annual taxa was recorded in the impaired relevés (19%) than in the minimally impaired (reference) relevés (2%), suggesting potential for metric development.
- Greater mean number, cover and Shannon-Wiener diversity of alien taxa were recorded in impaired than in reference relevés. In total 19 alien species were recorded in impaired and six in reference relevés. Three of those aliens (*Lythrum hissopifolium*, Medicago polymorpha** and *Samolus valerandii**) in reference relevés were not recorded in the impaired relevés. Typically only a single alien species was recorded in any reference sample whilst two or more alien species were recorded in 10 out of the 22 impaired relevés. Although the magnitude of difference in the number of alien species is too small to justify the development of a metric, the large difference in cover of alien taxa (<1% in reference *vs* 21% in impaired) does suggest that this may turn out to be a useful metric for phytoassessment purposes.
- Greater mean cover of woody taxa was recorded in reference (18%) than in impaired (5%) relevés; only alien taxa presented woody cover of greater than 2% in impaired relevés and only indigenous taxa representing all woody cover in the reference relevés. Within the reference relevés, nine indigenous woody taxa were recorded whilst in impaired relevés, five indigenous (one of which was shared with the reference relevés) and four alien woody taxa were recorded as listed in Table 5.9.

- The large difference in cover of indigenous woody taxa between disturbance categories (as averaged per relevé) suggests a useful metric for phytoassessment purposes (>18% in reference *vs* <5% in impaired).

Table 5.9: Woody species in supralittoral Fynbos relevés from Kenilworth

Reference species	Impaired species
Aspalathus sericea	*Leptospermum* scoparium**
Berzelia abrotanoides	*Paraserianthes* lopantha**
Erica margaritaceae	*Passerina corymbosa*
Erica sp.	*Senecio halimifolius*
Lachnaea uniflora	*Sesbania* punicea**
Passerina corymbosa	*Stoebe capitata*
Psoralea pinnata	*Stoebe* cf. *fusca*
Searsia laevigata var *laevigata*	*Thesium rariflorum*
Stoebe plumosa	*Xanthium* strumarium**

An asterisk at the end of a genus mark genera that are alien to the Fynbos biome.

- Indigenous sclerophyllous shrubs, the fine-leaved group of plants that typify Fynbos and from which its name is derived, had very low cover when present in impaired relevés: *Passerina corymbosa, Stoebe capitata, Stoebe* cf. *fusca* and *Thesium rariflorum* were recorded in impaired relevés with a median of 8% cover. A single alien sclerophyllous species, *Leptospermum scoparium**, had nearly 90% cover in a single impaired relevé. Six indigenous sclerophyllous species (*Aspalathus sericea, Berzelia abrotanoides, Erica margaritaceae, Erica.* sp., *Passerina corymbosa* and *Stoebe plumosa*) were recorded in reference relevés with cover values ranging from 2 to 70%. The difference in mean cover of indigenous sclerophyllous shrubs between reference (13%) and impaired (<1%) relevés suggests a potential metric.
- Four species of indigenous leafless graminoids (*Elegia nuda, Ischyrolepis paludosa, Restio quinquefarius* and *Restio burchellii*) were recorded in reference relevés, whilst two other species (the restio *Elegia tectorum* and the sedge *Ficinia nodosa*) were only recorded in impaired supralittoral Kenilworth relevés. The cover of leafless graminoids in reference relevés (14%) was considerably greater than that in the impaired relevés (1%); suggesting a potential metric.
- Lastly, greater mean number and cover of alien graminoid taxa were recorded in impaired than in reference relevés. The mean cover of alien graminoid taxa was considerably smaller in reference (1%) than in impaired (18%) relevés and is therefore a potentially useful metric for the Kenilworth supralittoral habitat.

5.3 Discussion

Vegetation of the littoral and supralittoral habitats was significantly and considerably different when examined at this finer spatial scale than previous analyses. Greater phosphate levels were recorded in impaired than in minimally impaired wetlands of the Cape Flats as is consistent with results of previous wetland assessment studies (Adamus *et al.* 2001, Malan & Day 2005b). Anthropogenic disturbance levels had a significant correlation to the floristic community differences between disturbed and minimally disturbed vegetation samples. These differences are most apparent in units of vegetation associated with a single terrestrial vegetation type and sampled within the same geographical locality. Characteristic species and diversity measures were developed as metrics or bioindicators of the difference between minimally impaired and impaired vegetation samples. These bioindicators were only developed from those vegetation units within which most distinct separation was apparent between impaired and minimally impaired vegetation samples of the same habitat, vegetation type and locality. More annuals and aliens and fewer woody taxa, and more Fynbos specialist plants such as taxa of the Restionacea and Ericacea, were found in impaired than in minimally impaired wetlands. These bioindicators are consistent with anticipated vegetation response to anthropogenic disturbances recorded in previous phytoassessment development studies (US EPA 2002b) other than for woody vegetation (Middleton 2002), which shows a trend that is specific to the Fynbos Biome. These bioindicators are specific to localities and to units of vegetation associated with specific terrestrial vegetation types and little generalisation is possible between different localities and vegetation types.

5.3.1 Hydrological differences

Within homogenous units of wetland vegetation, within the local landscape scale, differences are apparent in the floristic community structure of supralittoral and littoral habitat. Observed affinity of species for different hydrological habitat necessitates searching independently within each habitat for species that are discriminatory between reference and impaired environmental conditions. This corroborates the situation of hydrological zonation in the vegetation of many wetlands in North America (US EPA (2002b).

The identification of species discriminating between the littoral and supralittoral habitats was not the intent of the present study but could be performed with the present data set. The adoption of the Braun Blanquet sampling method of targeted sampling of

representative and homogenous stands of vegetation suggests that only a limited number of the full spectrum of plants along the gradient of hydrological change were captured in the relevés of the present study. No attempt was therefore made to determine discriminatory species between littoral and supralittoral habitats as this would best be performed with data captured along transects encompassing the full gradient of hydrological change (*sensu* Ellery *et al.*2003).

5.3.2 Community differences

Whilst PERMANOVA suggested significant differences existed between the community structure of reference and impaired relevés of Strandveld-associated and Fynbos-associated wetlands, ordinations revealed further spatially related differences (Section 5.2.2). Within a given hydrological habitat differences in community structure within the wetland relevés associated with a terrestrial vegetation unit are thus related to spatial and to apparently anthropogenic influences (Section 5.2.2). Spatially related variables such as climate and geology are therefore significantly correlated with community structure even amongst relevés and wetlands associated with a given vegetation type but a different locality (Lotus *vs* Kenilworth) (Section 4.3.3). Removing the natural variability caused by spatial (location) differences reduces natural variability in the reduced data set and increases the ability to focus on those community differences that correlate specifically with anthropogenic drivers of difference.

5.3.3 Discriminatory environmental differences

Human disturbances in the Kenilworth Racecourse area include land uses causing water loss in wetlands, poor water quality as a result of stormwater, livestock effluent and ablutions (a quarantine station) and potentially also due to past and present fertilizer use, and physical disturbances including road fill material, mowing and excavation. The lack of significant community differences between the littoral samples at Kenilworth may have been the result of patchiness in the human disturbance or landuse. Anthropogenic variables, including elevated phosphates in the water column as an indicator of eutrophication, have the highest canonical correlation to community structure differences between reference and impaired relevés. The land use surrounding and within impaired wetlands, and the known association of such disturbance impacts with environmental degradation in other wetlands around the world (e.g. Adamus *et al.* 2001, Middleton 2002, US EPA 2002a, Fore 2003, Clarkson *et al.* 2003, Dahl 2004) suggest that these anthropogenic variables are very plausible drivers of the observed differences in

community structure. Assuming that differences in community structure observed in reference relative to impaired relevés correlate with (or can be attributed to) anthropogenic influences, discriminatory attributes of the vegetation have potential value as metrics for phytoassessment. From the data presented here, we can assume this to be true for the supralittoral habitat associated with Sand Fynbos in the vicinity of Kenilworth.

Distance linear modelling (DistLM) could have been used in place of the more exploratory CAP to determine the linear combination and significance of environmental variables that best explain the observed community structure differences between impaired and reference wetlands. The DistLM process provided very similar results for the present data set with no meaningful alteration of the fact that anthropogenic influences are still the most highly correlated with, and thus considered important in influencing, difference between the reference and impaired wetlands of the present study (Corry in press).

5.3.4 Characteristic vegetation attributes

The influence of spatial autocorrelation that would exist within each locality may well increase the apparent significance levels of any t-tests of the difference of cover with which a given species was recorded between reference *vs* impaired relevés. This eventuality was reduced by using type 1 sum of squares and permutation of residuals under a reduced model for the generation of confidence levels (significance values) (Anderson *et al.* 2006). The discriminatory vegetation attributes between the reference and impaired supralittoral Kenilworth habitat are thus considered relatively robust as indicators of environmental difference.

The use of each species in an index does not pose the problem of collinearity. The combination of all of the potential metrics into a single index would, however, result in collinear variables being included or double counting and thus cause overestimation of the score and related state of environmental condition. For instance a metric based on alien graminoid taxa should not be used in conjunction with the cover of all alien taxa as this would lead to double counting. Similarly a metric based on sclerophyllous taxa should not be used in conjunction with a metric based on woody taxa as the former are also woody taxa and using both would again lead to double counting.

Internationally, dominance of woody species has been shown to be increasing in wetlands as a result of fire suppression and/or a combination with changes in grazing,

mowing, harvesting or other consumptive type activities or influences (see Middleton 2002 for a list of the many studies that have recorded increased woody and shrubby vegetation). Such influences may well be at work in an urban area such as Kenilworth where vegetation utilization is minimal, no large-scale grazing by ungulates occurs and fire regime is controlled due to the active management and conservation of the remnant patches of Acid Sand Plain Fynbos. In the Strandveld associated wetlands assessed at Driftsands, grazing, fire and harvesting of reeds, grasses and firewood were recorded anthropogenic disturbances and very limited shrubby or woody vegetation was apparent in either reference or impaired wetlands. More woody vegetation was recorded at the Lotus Strandveld-associated wetlands which, relative to the Driftsands wetlands, were less exposed to firewood collection and grazing by ungulates. In the case of the Fynbos vegetation, which relies on a cyclical fire regime and is considered naturally to contain a large proportion of shrubby or woody sclerophyllous vegetation (Cowling *et al.* 1997, Rebelo *et al.* 2006), depressed numbers and cover of woody and perennial taxa and elevated cover of herbaceous and annual taxa would be anticipated under disturbed conditions such as excessively frequent fires or other highly consumptive impacts on the vegetation. This is contrary to the phytosociological relationship of herbaceous and shrubby plants in the mountains of the Maloti-Drakensberg in southern Africa in which the presence of shrubs in wetlands is reported to indicate disturbance (Sieben *et al.* 2010).

Both reference and impaired sites of the present study were sampled from Kenilworth Racecourse, which is actively managed for conservation, and thus both have similar levels of consumptive vegetation influences, other than a single extreme case where the wetland is mowed and shrubby vegetation is entirely absent. The data collated in the present study suggest that wetlands associated with Cape Flats Sand Fynbos naturally support a contingent of shrubby vegetation and that physical disturbance reduces the prevalence of woody vegetation, increasing the prevalence of annual, alien and herbaceous plant growth forms. This begs the question as to whether the recorded increase of shrubby vegetation in the conserved sedge meadows of America and the fens of Europe (Middleton 2002) is not an indication of the absence of anthropogenic perturbation resulting in a return to a more naturally shrubby vegetated state?

5.4 Phytoassessment metrics for the Cape Flats

Given the vegetation attributes that were discriminatory between reference and impaired relevés in the Kenilworth supralittoral data, the potential does exist for the development of

phytoassessment tools for Sand Fynbos associated wetlands on the Cape Flats. A number of potential metrics derived for the supralittoral habitat of wetlands associated with the Cape Flats Sand Fynbos at Kenilworth in this study (Section 5.2.4) are presented in Table 5.10. The ratio of alien to indigenous taxa, observed to be an important discriminator between disturbance categories in Section 3.3.4, was not employed as a metric due to the more focused metrics (that do have affiliation to the components of this ratio) developed specifically for Kenilworth Fynbos. These metrics were based upon characteristic vegetation attributes, both species and life-history groups, which represented discriminatory cover values between reference and impaired wetlands. An arbitrary selection of the species, suggested to be discriminatory in Figure 5.5, were included here as potential metrics and any of the discriminatory species may suffice as metrics. The life-history groups present ecological generalizations as represented by numerous species and as such may have broad spatial applicability if such patterns are found to be common in numerous wetland vegetation units. The combination of these metrics in an index of environmental condition (or of biological integrity *sensu* Wilcox *et al*. 2002, Miller *et al.* 2006, Rothrock *et al.* 2008) could be achieved by scoring each metric depending on its affiliation to disturbance. The number of metrics included in such indices is recognized as influencing the accuracy with which environmental condition can be identified and the inclusion of 7 to 12 metrics is recommended (US EPA 2002a & b). The use of numerous metrics reduces the possibility that the inertia shown by some plant taxa (particularly woody taxa) will result in an inaccurate assessment of present environmental conditions. The inclusion of species with different life-history strategies also reduces the inaccuracy that vegetation inertia may cause with long-lived taxa and late-seral alien taxa. The inclusion therefore of a metric based on annual taxa would reduce the impacts of vegetation inertia on accuracy of phytoassessments.

The cover/abundance of many of the life-history groups are perhaps more intuitively simple to identify than particular species for non-specialists. For instance woody taxa and sclerophyllous woody taxa are easily identifiable with limited botanical training. Identification of further groups of taxa, which are easily identifiable for non-specialists (such as lawn *vs* bunch or tussock grasses), that have discriminatory potential between reference and impaired wetlands would greatly increase the ability of non-specialists to perform phytoassessments. Such a goal is necessary in South Africa in which a considerable need exists for the ability to assess wetlands but in which few specialists are trained by the conservation or legislative authorities (DWAF 2004).

Table 5.10: Potential phytoassessment metrics for an Index of Biological Integrity for supralittoral Cape-Flats-Sand-Fynbos-associated wetland habitat. A "*" denotes alien species.

Species / attribute	Reference	*vs.*	Impaired
Species metrics			
Typha capensis	Not present	*vs.*	≥12.5% average cover/sample
Cyperus textilis	Not present	*vs.*	88% typical cover/sample
Juncus capensis	≤10 specimens to 5 - 12.5% typical cover/sample	*vs.*	Not present
*Phalarais aquatica**	None or <5%	*vs.*	≥12.5% typical or average cover/sample
Cyperus sphaerospermus	≤10 specimens to 12.5% typical cover/sample	*vs.*	>12.5% typical or average cover/sample
Cynodon dactylon	None or <5%	*vs.*	≥5% typical or average cover /sample
Restio quinquefarius	≥5% average to ≥18% typical cover/sample	*vs.*	Not present
Life-history group metrics			
Annual taxa	≤5% cover/sample	*vs.*	≥18% cover/sample
Alien taxa	≤5% cover/sample	*vs.*	≥18% cover/sample
Alien graminoid taxa	≤5% cover/sample	*vs.*	≥18% cover/sample
Woody-indigenous taxa	≥18% cover/sample	*vs.*	≤5% cover/sample
Woody sclerophyllous shrubs	12.5-25% cover /sample	*vs.*	<5%
Indigenous leafless graminoids	>12.5% average cover/sample	*vs.*	<5% cover/sample

5.4.1 Validation of metrics

The targeted (vs. random) site choice used in this study is considered by some researchers to reduce the inferential power of this type of data set (see Fore 2003), suggesting that metrics developed for one set of wetland may not be applicable in any other wetlands or wetland vegetation stands within the same area. A targeted sampling approach, however, is used extensively for phytoassessment metric development across North America (US EPA 2002a & b), where a pragmatic approach to testing metrics is adopted. Testing is performed by checking, within wetlands not used in the development of the metrics, whether the metrics prove to be robust indicators of difference between vegetation samples from *a priori* determined reference and impaired categories of human disturbance. This should be done in wetlands of a similar HGM and vegetation type to those used in the development of the metrics. A suggested approach is to split the sampling data during phytoassessment development and develop metrics from one half and test them on the other half of the data set. This was not done in the current study due to insufficient comparative samples as a result of the unforeseen extreme

heterogeneity of the flora of wetlands shown by this study to exist within the Western Coastal Slope region.

5.4.2 Spatial applicability of metrics

Considerable difference between the vegetation communities of wetlands of different phytogeographical areas restricts the potential that metrics developed and tested for one geographical region will work in other areas. For instance the significant differences in vegetation communities between the localities assessed within the mediterranean-region of the Western Coastal Slope suggests that for Sand-Fynbos-associated wetlands it is unlikely that metrics for supralittoral habitat at Kenilworth will work in supralittoral habitat at Lotus River or in other sub-regions of the Cape Coastal Lowlands. Testing the supralittoral-Kenilworth Fynbos metrics in a different vegetation unit such as the Driftsands Dune Strandveld samples is unlikely to reveal any clear answers due to the different characteristic species associated with Strandveld and Fynbos vegetation. The intermingling, or limited difference, of the community structure of impaired and reference Strandveld relevés at Driftsand (see ordination in Figure 5.1) also suggests that the metrics developed for Kenilworth will not be useful to differentiate between disturbance categories of these Strandveld relevés. As the only reference relevés associated with Sand Fynbos were sampled in the Cape Flats it is also not possible to use the Sand Fynbos relevés from Hopefield, Atlantis and Leipoldtville of the West Coast sub-region to test the metrics developed from the supralittoral Cape Flats relevés.

Within the USA, where there has been considerable effort spent on phytoassessment development, metrics are developed and tested independently for each ecoregion (US EPA 2002b) as these regions have different vegetation. Species with broad environmental tolerances and resultantly broad distributions, and life-history groups that have consistent responses to disturbance, both suggest some potential for metrics with geographically broader applicability than was apparent within the present limited data set and that may be useful across numerous phytogeographical regions.

5.5 Conclusion

The limited spatial extent for which the supralittoral-Kenilworth-phytoassessment metrics would be applicable suggests that considerable baseline data would be required to develop phytoassessment for wider use in the Fynbos Biome. The phytoassessment metrics developed from the supralittoral-Kenilworth data set may well have wider spatial

application than the Kenilworth-Sand-Fynbos wetlands. The limited geographical range incorporated by the samples within the present study perhaps artificially over-emphasises the difference of wetland vegetation associated with different localities within a given unit of terrestrial vegetation such as the Cape Flats Sand Fynbos. Differences between the Lotus and Kenilworth relevés of this vegetation type certainly reflect climatic and geological differences. Sampling vegetation from the full spatial extent of the Sand Fynbos associated wetlands of the Lotus, Kenilworth and any inter-leading localities may reduce the differences observed in the present data set. There is therefore potential to determine phytoassessment metrics with broader geographical application in the Fynbos biome than these metrics expounded above. This development would rely on collation of considerable baseline data such as greater accuracy in the geographical extent of the distribution of wetland species and of the phytosociology of the associated life-history groups. The metrics developed for the supralittoral Sand Fynbos wetland habitat in the present study represent user-friendly phytoassessment metrics for the Fynbos Biome of the Cape Floral Kingdom.

The lack of replicate reference and impaired samples in each of the phytogeographically different units of wetland vegetation, i.e. those associated with different terrestrial vegetation units, is an obvious flaw in the sampling design of the present study. This resulted in the inability to search for differences between impaired and reference relevés, and thus the inability to search for metrics, for wetlands associated with many of the terrestrial vegetation types sampled in the present study. This flaw resulted from the unfortunate acceptance of the theory proposed by Walter (1973), and further promulgated by Mucina *et al*. (2006a), that wetland vegetation is azonal.

To the best of my knowledge the approach adopted in this thesis represents the first attempt to prove strong correlation between anthropogenic disturbances with wetlands categorized as being impaired (Section 5.2.3) before searching for vegetation attributes (metrics) that distinguish between reference and impaired wetlands. Previous studies of this nature have adopted the approach that significant levels of human landuse imply impairment of environmental condition but have not attempted to test this assumption with empirical evidence. This departure from the standard approach represents a more empirically defendable argument for the adoption of phytoassessment metrics as representative measures of environmental conditions.

CHAPTER 6

DISCUSSION, CONCLUSIONS AND RECOMMENDATIONS

This study has investigated wetland phytogeography in the Fynbos biome and the potential of wetland plants to assist in the assessment of wetland environmental condition. The ability to assess wetland condition will assist in the application of the National Water Act (Republic of South Africa 1998) and in conservation planning and management. A number of findings have emerged from the present research about the phytogeography of wetland macrophytes. Most notably, it is apparent that there is considerable natural variability in macrophyte distribution, which seems to be hydrologically driven within wetlands, whilst, within regions, it seems to be driven by climatic and geological differences. These climatic and geological discriminators are the same as those which correlate with the distribution of different units of terrestrial vegetation (Rebelo *et al.* 2006) and hence suggest that wetland vegetation is no more azonal than upland or terrestrial vegetation units in the coastal lowlands of the Fynbos biome. The European Union recognises broad climatic zones of wetland vegetation within which, geological substrates are a secondary discriminator of different units of wetland vegetation (Eur15 1999). In the United States of America separate units of wetland vegetation are based on ecoregions with climatic and geological discriminators (US EPA 2002b) and on the Pacific coast of North American the work of Peinado *et al.* (2007) also suggests climatically zonal distribution of wetland vegetation with some edaphic determinants of reportedly azonal distribution. The distribution of wetland vegetation of the Fynbos biome appears to be similarly constrained by intrazonal climatic and edaphic determinants.

The similarity in spatial distribution of wetland and terrestrial units of vegetation in the coastal lowlands of the Fynbos biome suggests that the associated terrestrial vegetation units present a means of classifying distinct homogenous units of wetland vegetation. Examination of the vegetation within these units identified useful phytoassessment metrics for discriminating between reference and impaired wetlands. This separation of naturally and anthropogenically distinct units of vegetation provides a general framework for development of metrics for the phytoassessment of wetlands in all biomes of South

Africa. The wetlands of the Cape coastal lowlands examined in this work are a heterogeneous microcosm of the wetland habitat diversity in South Africa that may well be able to inform our understanding of the macrocosm. The considerably greater diversity of vegetation within the Fynbos biome than other biomes in South Africa suggests that outside of this biome broader areas of wetland vegetation may be homogenous and that phytoassessment development may thus be less restricted to small spatial scales. In this chapter these findings are discussed along with the application of phytoassessment metrics. Avenues of potential future research are suggested that may increase our understanding of the phytosociology of wetland plant communities, and the impacts of anthropogenic disturbance, and thereby inform the effort to develop more accurate phytoassessment tools.

6.1 Wetland vegetation in the Fynbos biome is not azonal

The proposed intrazonality (or occurrence exclusively within a climatic zone) of wetland vegetation units within the Fynbos biome (Mucina *et al.* 2006a) (Section 1.4.4) suggested initial cognisance of the zonal differences of wetland vegetation between biomes. Yet within intrazonal wetland vegetation units Mucina *et al.* (2006a) proposed a broad distribution of wetland macrophytes that was not considered to be constrained by climatic and edaphic parameters that constrain intrazonal units of terrestrial vegetation. This proposed broad and "azonal" distribution was, rather, considered to be driven by hydroregime and salt concentrations exerting an influence greater than the macroclimate (Mucina *et al*. 2006a). My research has shown that within the Fynbos biome wetland phytogeographical regions (Section 3.3.3) correlate with the spatial variation of geology and climate (Section 4.3.2) in a similar manner to the intrazonal terrestrial units of vegetation, therefore challenging the supposed azonality of wetland vegetation. This suggests that, within the Fynbos biome, the distribution of wetland vegetation is no more or less affected by soils and climate (or azonal) than that of the (intra)zonal terrestrial vegetation.

That climatic and geological variables correlate with the phytogeography of wetland species was made apparent by the significant environmental differences both between and within sub-regions of the Cape coastal lowlands (Section 4.3). These climatic and geological drivers also correlated with the floristic community structure differences evident as beta diversity variation in the Cape Lowland Freshwater vegetation unit of Mucina *et al.* (2006a) across the mediterranean-region of the Western Coastal Slope

(Cowan 1995) of the Fynbos biome (Section 3.3.3). Whether the littoral or supralittoral vegetation is used to describe the wetland plant community, contrary to the theory of Walter (1973, 1985 as used by Mucina *et al.* 2006a), these communities are no more azonal than the terrestrial vegetation, and the distribution of their species is generally not broad.

Relative to neighbouring terrestrial habitat, elevated water availability in wetlands alters the microclimate of the ground in which wetland plants are rooted. Ambient temperature and evapotranspiration affect the amount of water available, however, and have a "zonal" influence on both terrestrial and wetland plants. In permanently inundated habitats, such as situations in which the influence of upwelling groundwater is significant, the zonal influence of evapotranspiration will be of lesser importance in determining the hydroregime than in seasonally inundated or saturated wetland habitats. The majority of the wetlands assessed in the present study were seasonal and supplied by rainfall and surface water flow rather than groundwater. Similarly water availability and landform affect nutrient concentration as a result of influx, leaching and evaporative concentration of salts; but under natural conditions the availability of these salts is predominantly determined by the underlying geology, which has a zonal influence on vegetation. A possible anomalous situation exists for salt-laden intermittent rivers, which may transport salts to an area other than that from which the salts are derived, thereby perhaps creating an azonal macro-ecological driver of saline soils surrounded by and overlaying the zonal geology and thereby create an azonal unit of vegetation.

The vegetation of the littoral habitat of wetlands with seasonal to permanent inundation does, however, contain some (sub)cosmopolitan species (e.g. *Typha capensis*) as a result of the influence of hydroregime on increased water availability. Such (sub)cosmopolitan distribution is perhaps more determined by species with broad environmental tolerances than by the existence of like habitats. For instance in seasonal and permanent freshwater wetlands *Typha capensis* grows in the supralittoral and littoral habitats of both nutrient poor and nutrient rich soils and occurs all over Southern and Tropical Africa (Goldblatt & Manning 2000). The influence of a wetland's hydroregime essentially creates habitat and resultant vegetation different from the surrounding terrestrial vegetation. It is hypothesized that the distribution of wetland vegetation is influenced by geological and climatic differences between areas within a biome and that it is therefore inaccurate for wetland vegetation to be considered azonal. Rather the wetland habitat and its vegetation should be considered as anomalous relative to

surrounding terrestrial habitat and vegetation. That wetland vegetation of the Fynbos biome is not azonal, but determined by the same factors of climate and geology that determine the distribution of terrestrial vegetation, is a key finding of the present study. Within the Fynbos biome, therefore, both wetland and dryland (terrestrial) units of vegetation are essentially intrazonal units of vegetation with similar constraints on their distribution. Description of dryland or wetland units of vegetation as zonal or azonal is considered inaccurate as both are contained within the Fynbos biome and are thus both intrazonal *sensu* Walter (1973, 1985).

The outcome of this finding is that the wetland vegetation within the mediterranean-region of the Western Coastal Slope does not represent a single phytogeographical region as was postulated by Cowan (1995). Equally, the Cape Lowland Freshwater (CLF) vegetation unit of Mucina *et al.* (2006a) is not a homogenous phytogeographical entity with similar species across the entirety of the Cape coastal lowlands. The fact that the wetlands from each terrestrial unit of vegetation (and even every locality within each unit) exhibited significantly different communities of wetland plants suggests the existence of considerable natural variability, which within a conglomerate of vegetation units, masks community differences between reference and impaired wetlands. This necessitated searching for potential metrics for phytoassessment within each separate locality even when working within wetlands associated by a common terrestrial unit of vegetation (Section 5.2.3 and see Section 6.3).

6.2 The importance of hydrological zonation within wetlands

An important consideration realized during initial sampling attempts, using the vegetation sampling protocol for phytoassessment used by the US EPA (2002b), was that the wetlands of the Cape coastal lowlands are stratified into hydrologically determined concentric zones. The influence of such zonation on habitat availability is considerable and separates significantly different vegetation communities into littoral and supralittoral habitats (Sections 1.5.2, 3.2.2 and 5.2.1). This suggested that the single-large-plot sampling methods suggested by the US EPA (2002b) would not sufficiently capture the heterogeneity of habitats found in the wetlands of the Fynbos biome and that numerous smaller plots would be required to do so to avoid masking natural variability. The Braun Blanquet vegetation sampling protocol facilitated sampling of homogenous vegetation stands considered characteristic of each of the supralittoral, littoral and aquatic hydrological zones. Differences in community structure between supralittoral and littoral

vegetation was inconsequentially small when examined at the biome-scale or regional-scale incorporating the whole data set (Section 3.3.2). At the local landscape scale, when examining community structure differences within wetlands associated with a given unit of terrestrial vegetation, however, significant differences were apparent between supralittoral and littoral vegetation (Section 5.2.1). Differences in the vegetation of different hydrological habitats are apparent in the wetlands of the world as recognized by the work of numerous researchers (Eur15 1999, Middleton 2002, US EPA 2002b, Brock 2003, Clarkson 2004, Declerk *et al.* 2006) and in southern Africa by the work of Kotze & O'Connor (2002), Ellery *et al.* (2003) and Sieben *et al.* (2010). Within the USA, the vegetation of the littoral zone (emergent vegetation) has been the focus of much phytoassessment development in depressions (US EPA 2002b), whilst the supralittoral zone has been the focus for assessing the condition of wetland meadows (Middleton 2002).

It is apparent that assessments of vegetation type or condition are influenced by the hydrological zones in which samples are made. Wetland vegetation habitat classification, and/or the development of phytoassessment metrics, and the outcome of phytoassessment, must therefore take into consideration both the influence of hydrological zonation and the full complement of habitats within wetlands.

6.3 Phytogeography of distinct wetland vegetation units

International approaches to wetland phytoassessment suggest that metrics need to be developed independently for regions that prove to have different vegetation communities (Section 1.4). Uncertainty about the areas of South Africa that contain comparable communities of wetland vegetation, and the use of Cowan's (1995) wetland regions in the classification of wetland vegetation types by Mucina *et al.* (2006a), led to investigation of the homogeneity of wetland vegetation within one of these essentially untested wetland regions. All effort was focused within the mediterranean-region of the Western Coastal Slope (Cowan 1995) rather than spreading the sampling over many regions that would encompass many more natural differences in terms of climatic and geological drivers of phytogeography than existed within this single wetland region. Within the Western Coastal Slope wetland region (Cowan 1995), sampling of inland, isolated freshwater wetlands was performed in a number of localities, each within one of three sub-regions, to determine whether all of these wetlands held comparable plant communities. Should all of the wetlands have proved comparable, then the Western Coastal Slope would have been considered a single phytogeographical region and, as such, a region that would

suffice for the purposes of development and application of phytoassessment metrics. The outcome of this present study revealed that, within the Western Coastal Slope, different wetland plant communities existed within different geographical areas. The spatial scale of these differences was relatively small occurring within tens of kilometres as a result of geological substrate and climatic differences. These differences follow a similar spatial pattern as that reflected by the terrestrial (or intrazonal) units of vegetation (*sensu* Rebelo *et al.* 2006).

- Wetlands of the Cape Flats have significantly different plant communities from those to their south east on the relatively more arid Agulhas Plains, or the considerably drier West Coast to the north of the Cape Flats.
    - Within the Cape Flats, the most comprehensively sampled of these above three sub-regions of the Western Coastal Slope:
        - lowland freshwater wetlands of the same HGM type have significantly different vegetation if, for instance, they are from acidic or alkaline substrates; and
        - Wetlands associated with the same terrestrial vegetation unit (suggesting similar environments and particularly geological substrates) but from different localities can still have significantly different wetland vegetation communities (for details see Section 5.2.2 and 6.3.1).

These differences between wetland vegetation of each locality within the Western Coastal Slope suggest that none of the currently defined wetland regions (Cowan 1995), ecoregions (Kleynhans *et al.* 2005), bioregions (Brown *et al.* 1996 and / or Rutherford *et al.* 2006) or even terrestrial vegetation units (Rebelo *et al.* 2006) provide homogenous units of Cape Lowland Freshwater vegetation. All of these phytogeographical units contain a degree of natural variability that may mask anthropogenic variability and none thus provides a unit within which it unquestionably makes sense to attempt the development of metrics for phytoassessment purposes. The homogeneity of vegetation within any of these units should be assessed for natural variation before attempting to develop phytoassessment metrics. As described below, however, wetlands associated with a given terrestrial vegetation unit can be expected to be more similar to each other than to wetlands associated with a different terrestrial vegetation unit.

6.3.1 Wetlands with similar geological substrate

In the Cape coastal lowlands of the Fynbos Biome the terrestrial vegetation units of Rebelo *et al.* (2006) could potentially provide a surrogate for determining homogenous units of wetland vegetation. Yet, even within terrestrial vegetation units, differences were apparent between the wetland plant communities sampled in different localities (Kenilworth *vs* Lotus). The Kenilworth and Lotus River floodplain wetlands are situated within Cape Flats Sand Fynbos terrestrial vegetation and yet each locality holds significantly different wetland vegetation communities. A spatial gradient in both climatic and edaphic parameters was strongly correlated with these differences in species distribution and community structure (Section 4.3.3). Natural ambient climatic and geological environmental differences were also shown to exist between Strandveld and Fynbos associated wetlands of the Cape Flats (Section 4.3.3). Significant natural differences between the wetland plant communities of separate localities but the same terrestrial vegetation unit reduces the ability to identify phytoassessment metrics that would be applicable for all wetlands associated with a given terrestrial vegetation unit. Within localities in which reference and impaired wetlands from the same geological substrate were compared, significantly different vegetation communities were detectable between each disturbance category at each of the Cape Flats localities. However, a failing of the sampling design of this research was that insufficient replicates of both reference and impaired, littoral and supralittoral vegetation were sampled at each locality and even, in some cases, associated with each terrestrial vegetation unit.

Differences between Lotus and Kenilworth Sand Fynbos wetlands, may have been due to the wetlands in each of these localities acting as a source of propagules for other local wetlands, thereby entrenching local similarity (Aznar *et al*. 2003). The localities assessed in this study were focused in areas with large wetlands or conglomerates of many wetlands. Distances between vegetation communities of the same type are known to influence similarity, partially due to propagule dispersal ability, but also due to environmental differences (Cowling *et al.* 1992, Cowling *et al.* 1996, Latimer *et al.* 2005). Fragmentation, as a result of increased distance between wetlands, is known to impact on community composition and adaptability to human alteration (Adamus *et al.* 2001). Assessment of other wetlands, between the Kenilworth and Lotus foci, but still within Cape Flats Sand Fynbos vegetation, may reveal a continuum of similar wetland vegetation that was not apparent due to the spatially restricted foci of the present study. Such an eventuality would suggest that wetlands associated with the Cape Flats Sand Fynbos terrestrial vegetation unit do in fact present a single phytogeographical unit for

phytoassessment purposes. Testing this hypothesis was beyond the scope of the present project but is an interesting topic for further study.

The wetland vegetation groups identified by the National Freshwater Ecosystem Priority Areas (or NFEPA) project (Roux *et al*. 2006) (Section 1.4.3) are a network of phytogeographical units within each of which considerable phytosociological similarities exist. These similarities exist due to the incorporation, in each NFEPA vegetation group, of wetlands with similar geological substrates as linked by similar terrestrial vegetation units. Further testing of these groups is advised in the light of the differences evident between wetlands associated with each of the terrestrial vegetation units in the South West Sand Fynbos NFEPA group (Section 3.4.3. (i)). Beyond the *Capensis* and *Drakensbergensis* phytochoria these NFEPA groups may well be more homogenous as explained below.

6.3.2 Phytogeographical regions beyond the Fynbos lowlands

The Western Coastal Slope is part of the Cape Floral Kingdom or of the *Capensis* phytochorion (a phytogeographical unit of land based on the striking presence and absence of groups of plant taxa). For terrestrial vegetation, the *Capensis* and *Drakensbergensis* floral regions are recognized areas of high floral endemism and diversity. Given the diversity shown to exist within the wetland plant communities of the Western Coastal Slope in this study, it is safe to assume that the *Capensis* and *Drakensbergensis* phytochoria are both likely to contain many areas with naturally distinct wetland vegetation communities. Within the highlands of Mpumalanga, initial investigations also suggest that considerable natural environmental heterogeneity exists among isolated depressional wetlands in the Lake Chrissie area (Martin Ferreira, Pers. com. PhD candidate, University of Johannesburg, 2009). Regions that have greater spatial homogeneity in the environmental parameters that drive species distribution, than in the Western Coastal Slope, will probably require less subdivision into areas of distinct vegetation. The homogeneity of the terrestrial vegetation and environmental parameters within the western Free State suggests considerable homogeneity in wetland vegetation in this area of South Africa (Collins 2011).

6.4 Phytoassessment in other regions of South Africa

In South Africa phytoassessment has potentially broader applicability than the use of invertebrates for inferring environmental condition in wetlands since such methods are restricted to habitat that is inundated to at least ten centimetres in depth (Bird 2010). The arid to semi-arid climate of much of South Africa results in many ephemeral, seasonal and even perennial wetlands that are, at their wettest, only ever saturated. Hence, phytoassessment has nation-wide potential, provided reliable region-wide metrics can be developed for homogenous phytogeographical regions of wetland vegetation.

6.5 Sampling Design: What worked *vs* what didn't work?

6.5.1 Human Disturbance Score

For phytoassessment development, a measure of human disturbance needs to be derived independently from biotic data in the target ecosystem in order to avoid simply choosing aspects of disturbance or measures of biology that match our expectations (Fore 2003). The extent of monospecific stands of indigenous or alien vegetation coverage within a target wetland were considered as biotic data representative of the wetland and thus were inappropriate for the assessment of the amount of human disturbance. The extent of monospecific stands of vegetation and "expert" judgement of habitat heterogeneity within wetlands were initially included in the measurement of the Human Disturbance Score (HDS) but they were subsequently removed to avoid circular reasoning or researcher expectation/bias. The extent of alien vegetation outside of the wetlands was, however, included as a measure of disturbance. These areas outside of the wetlands did not constitute an area within which vegetation was empirically assessed to measure wetland environmental condition.

Allied to the measure of soil and water chemistry that reflect eutrophic conditions the HDS proved reliable in its ability to assess the impacts and disturbances affecting wetlands and their immediate surrounds as a result of human landuse and associated activities. The inclusion of two additional factors could add to the accuracy of the HDS:

i. The roughness of the buffer zone of terrestrial vegetation around each wetland would provide an indication of the ability to reduce influx of sediments and associated nutrient load (Reddy & Gale 1994, Mitsch & Gosselink 2007). Whilst all wetlands in the present study were in relatively flat land forms, in sloped landscapes, the slope of the buffer zone may also be an important consideration. Correlations between buffer zone vegetation roughness, slope and sediment

influx have yet to be determined for South African wetlands but their importance is well established by considerable international research (Dr Macfarlane, UKZN, pers. com. 2011).

ii. Landscape and vegetation can both be structurally impacted by physical disturbance events or landuse. Separately scoring each of these would be useful in HDS determination. Only landscape impacts were scored in the present study. Impacts to vegetation should only be assessed in the 100 and 500 meter radii surrounding a wetland, as scoring those impacts within the wetland may lead to circular reasoning.

As an alternative to the HDS, the Wetland Index of Habitat Integrity (Rountree *et al.* 2007) or WET-Health (Macfarlane *et al.* 2008) could be used to rank environmental condition within a set of wetlands. Both of these tools use vegetation and impacts on vegetation as indicators of disturbance, however, thereby reducing their independence from the biological target for phytoassessment development purposes. Abiotic aspects of both of these tools were incorporated in the development of the HDS (Section 2.2.4).

6.5.2 Comparable samples

Comparability of samples depends on like habitat being sampled. A number of samples from littoral habitat in the present study were outliers from the littoral data set due to their maximum potential depth of inundation. This resulted in a number of impaired relevés with no comparable reference samples from similar depths (Section 5.2.2). Sampling a range of depths, or focusing on a given depth, in each wetland is therefore important as the maximum depth of annual inundation can determine the species and resultant vegetation community that can survive in a given location (e.g. Cook 2004).

Similar sampling design issues resulted in a number of disjunctions in the data set compiled for wetland vegetation associated with each terrestrial vegetation unit. There were for instance, unreplicated reference and impaired samples, substrate texture classes and HGM units. The theory that guided the sampling design was that wetlands were azonal (Walter 1973) and thus that the Cape Freshwater Lowland vegetation unit (Mucina *et al.* 2006a) would provide comparable samples across the Fynbos biome. As the intention of the study was to test the potential for phytoassessment development within this supposedly homogenous phytogeographical region little emphasis was placed

on ensuring that there were replicated samples within every location and every terrestrial vegetation unit within that region. Given the observed heterogeneity, for many areas, insufficient sample replication for phytoassessment purposes resulted from the sampling performed.

Apart from the identification of homogenous phytogeographical regions as determined by the macro-ecological drivers of climate and geology, sufficient replicate samples with comparable habitat as determined by water and soil depth, substrate texture, and a range of impairment are all necessary components of a sampling universe within which it should be possible to develop phytoassessment metrics. These environmental parameters thus provide a framework for determining comparable wetlands for phytoassessment purposes.

The different vegetation communities presented by herbaceous, scrub-shrub and forested wetlands are important biological divisions separating comparable vegetation units for phytoassessment purposes (US EPA 2002b) that are now also adopted for classification of different wetland habitats in South Africa (SANBI 2009). The co-occurrence of herbaceous and scrub-shrub vegetation in the wetlands of the Fynbos Biome sampled in the present study typically occurred in different hydrological zones with woody vegetation typically present only in the supralittoral habitat. In the Fynbos Biome these biological divisions are thus separated by hydrological parameters of depth of inundation and/or saturation that were also used as divisions between supralittoral and littoral hydrological habitats (Section 1.5.2). In other biomes greater cognisance may need to be taken in regard to using the biological divisions to classify separate units of vegetation within which to search for and/or test or apply phytoassessment metrics.

6.5.3 How many samples are enough?

Where differences were discernable between disturbance categories within hydrological zones of wetlands from each locality of a given terrestrial vegetation unit, an average of 20(±2) reference and 14(±3) impaired samples were compared. In localities in which no significant difference between disturbance categories was discernable, too few comparable vegetation stands were sampled. From this work it appears that a minimum of 40 reference samples and 30 impaired samples from a given hydrological zone of comparable wetlands from any single geological substrate would provide sufficient data to facilitate the determination and testing of metrics for phytoassessment. Differences in

community structure between the Kenilworth and Lotus Cape Flats Sand Fynbos associated wetlands suggest that in areas with considerable environmental heterogeneity, 40 reference and 30 impaired samples should be made in each hydrological zone of every locality where there is a concentration of wetlands.

6.5.4 Nutrient analyses

In the wetlands in which they were measured, water column nutrient concentrations were assessed as an average value per wetland following the recommendations of the US EPA (2002) and the research of Smith *et al.* (2007). Nutrient load ascertained from this data was useful for corroborating the HDS derived disturbance category each wetland was assigned to and for distinguishing units of similar habitat. Substrate (soil) nutrient data was assessed per vegetation sample rather than as an average value per wetland following botanical research convention (Kent & Coker 1992) and against the recommendation of the US EPA (2002b). The multiple soil samples per wetland revealed considerable intra-wetland variability in nutrient concentration (Corry in press). This revealed that more accurate environmental requirements (the autecology) of a given taxon can only be determined if nutrient data are measured where a vegetation sample is taken, rather than using a value from a single or even the average of a number of pooled samples per wetland. Investigation suggests that intra-wetland variation in water column nutrient concentrations is also apparent when multiple samples per wetland are tested (Dr G Ractliffe, pers. com. University of Cape Town, 2009). The homogenizing influence of water on these water soluble nutrients is, however, likely to reduce the extent of intra-wetland variability relative to that observed within the substrate.

Both water and soil chemistry were used in the determination of comparable wetlands and to search for evidence of the impacts of human disturbance using CAP (Section 5.2.3). The lack of a comprehensive set of soil and water chemistry measures for all wetlands of the present study makes it difficult to ascertain which is more important or informative about the impact of human disturbance and the phytosociology and autecology of any given species. Further research should be conducted in order to ascertain which of these sets of variables are of greater use for autecological purposes.

6.5.5 Univariate *vs* multivariate metric development

Examination of the distribution of single species (or of life-history groups) along single environmental gradients such as phosphate load may assist in the determination of

geographically more widely applicable metrics than was achieved with the ordination of community structure used in the present study. The use of scatter plots to examine macrophytic taxa distribution in response to univariate disturbance has been widely used for phytoassessment development (Gernes & Helgen 2002, Mack 2007) and is recommended by the (US EPA 2002b). This scatter plot approach is dependent on all samples being of comparable habitat (US EPA 2002a) and could thus have been used in the present study within the data sets representative of a given hydrological zone and location within a terrestrial vegetation unit. Community structure differences between reference and impaired conditions are, however, considered to provide more accurate representation of environmental conditions than single species (Magurran 2004). Phytoassessment metrics with good discriminatory potential are reportedly derived from species or other vegetation attributes ascertainable with the multivariate approach adopted in the present study (Fore 2003, Dahl 2004).

6.5.6 Species demographics to be sampled per wetland or vegetation stand?

Sampling characteristic and homogenous vegetation stands, following the Braun Blanquet protocol, facilitated the determination of an estimate of the cover and abundance of plants that occupied reference or impaired samples and could be averaged per hydrological zone, per wetland or per locality. The measurement of cover/abundance derived per relevé within a homogenous vegetation stand was far more informative than the weighted-average per hydrological zone that was developed from these samples. Averaged cover values for a given species in a given hydrological zone, considerably underestimated the typical cover of said species as found in sampled vegetation stands and yet overestimated the total found in the whole wetland. For instance using an average from all samples within a given hydrological zone, *Typha capensis* may occupy only 30% of a wetlands littoral zone, yet within stands in which it is found it will often occupy a median cover of 88%; and it may have a total wetland cover of only 10%. The concept of weighted-average used to determine the total cover of a species per wetland in the present study (Section 2.2.8.2), was based on the proportional wetland area (percentage) occupied by the hydrological zone that a sample was found within rather than the proportional area of the homogeneous stand of vegetation that it represented. Estimation of the total cover per wetland and/or per hydrological zone that a species represents would facilitate greater accuracy in determination of metrics. The way in which cover/abundance is sampled has important implications for the way in which metrics are scored for phytoassessment purposes (as described below Section 6.5.7).

6.5.7 Application of metrics in phytoassessment

In a phytoassessment, species and diversity metrics must be used to measure the same fraction of area within a wetland as the fraction from which they were derived in the development phase. The metrics outlined in Figure 5.5, Table 5.8 and Table 5.10 are all qualified by an area, be it sample, hydrological zone or wetland average, within which they represent different values for reference relative to impaired conditions. Metrics developed from average value per hydrological zone (**average x% / zone**) must be scored in a phytoassessment using average cover per hydrological zone. Scoring such metrics using the estimate of typical cover per sample (**typical x% / sample**) from only vegetation stands in which a species is recorded would result in considerable overestimation of the importance of the species or attribute used in the development of the metric ((**average x% / zone**) **<** (**typical x% / sample**)). Similarly scoring such a metric (**average x% / zone**) as the approximate cover of said species within the whole wetland (**average x% / wetland**) would lead to considerable underestimation of the importance of the metric ((**average x% / zone**) **>** (**typical x% / wetland**)).

6.5.8 Metric potential of single relevés with single-species dominance

Do average and typical cover of species that occur in more than one sample within any locality present more accurate comparative information than species only represented by a single sample? The graphic representation of typical cover of a species (e.g. of *Cyperus textilis* or of *Typha capensis*) that occurs in only a single sample, but with large cover, suggests considerable importance of such species (see Figure 5.5). Species representative of monospecific stands are often only sampled once in a survey as the single relevé would ably represent said vegetation stand. Such a species, however, is considered to have low "fidelity" being neither a "constant companion" due, for instance, to "*occurring in 60% of samples from a community*" nor a dominant species "*a constant companion with 25% mean cover*" (Westhoff & van der Maarel 1978). In the Braun Blanquet protocol, however, a stand of vegetation is only sampled if it is considered representative of the plant communities present at a site. Stands with 100% cover (88% median cover) by a single species (mono-dominant or monospecific stands), often represent an impacted state such as when invaded by an alien species or occupied by a very successful competitor for resources such as by *Typha capensis* or *Cyperus textilis*. Essentially such samples are indicative of conditions that are favourable for such species. In the approach of Westhoff and van der Maarel (1978) a species such as *Typha*

capensis that may occupy 10% of a wetland with 88% median cover, thereby possibly having the greatest total cover of any species within the wetland, does still have low fidelity relative to all other stands of vegetation in the wetland and is thus not considered dominant. When considered per hydrological zone, in which said species may occupy as much as 30% of the zone, however, the dominance and fidelity ratios are different. Fidelity and dominance are a construct of the sampling strategy. For phytoassessment, therefore, an estimate of the percentage cover that a homogenous stand occupies in a wetland, or a hydrological zone thereof, would assist with establishing the total cover that a species from that stand occupies in the entire wetland. And thereby be informative about that species dominance. Only if all of the different characteristic stands of vegetation within a wetland are surveyed can the fidelity of a species to different characteristic stands and hydrological zones be accurately determined. If a known disturbance effect or environmental condition is associated with monospecific stands of a given species, then this is more informative for phytoassessment than the fidelity of this species to other stands of vegetation within the wetland. Estimates of the total cover of mono-dominant species within a wetland or its hydrological zones would be of use in the development of metrics for phytoassessment purposes. Other than these two suggested additions, the Braun Blanquet sampling approach as presented by Westhoff and van der Maarel (1978) represents a very useful means of assessing wetland vegetation for the purposes of the development of phytoassessment.

6.6 Way forward

From the evidence and information provided by the present study it is clear that phytoassessment does have considerable potential for determination of environmental condition in wetlands of the Fynbos Biome and of South Africa. A number of suggestions are given in the following sections that could assist with the endeavour to address phytosociological understanding and phytoassessment needs and to utilize already collected wetland vegetation data.

6.6.1 Wetland Classification

Wetland classification can be done with many different aims. Phytoassessment requires the classification of wetland habitat units with minimal natural environmental variation such that impacts of anthropogenic disturbance would not be masked by natural environmental variability. The outcomes of the present study have revealed that discriminators used in the SANBI (2009) National Wetland Classification System to

separate different units of wetland largely fulfil this task for the purposes of phytoassessment other than the classification of different ecoregions and different hydrological zones. The following bullet points describe some of the outcomes of this studies examination of the discriminators and provide suggestions, where appropriate, to improve classification discriminators to assist phytoassessment development.

- The distinction between different units of wetland vegetation in the present study (Section 3.3.3) has revealed that the ecoregions of Kleynhans *et al.* (2005), the Bioregions of Rutherford *et al.* (2006), and the NFEPA wetland vegetation groups of Roux et al. (2006) do not distinguish homogenous regions of wetland vegetation in the Fynbos biome. Furthermore it is apparent the mediterranean Western Coastal Slope wetland region of Cowan (1995) also held considerable internal variability. All of these units are therefore considered too broad to represent useful units for classifying homogenous phytogeographic units for phytoassessment development. The considerable variation of vegetation with the Fynbos biome suggests that different edaphic substrates (as used for the separation of the terrestrial vegetation units of Rebelo *et al.* (2006)) are a necessary discriminator for classifying like units of wetland vegetation for phytoassessment purposes. The extent to which each of the terrestrial units of vegetation (*sensu* Mucina & Rutherford 2006) hold significantly different units of wetland vegetation needs to be tested outside of the Cape coastal lowlands that were tested in the present study.
- The separation of hydrological zones within wetlands into the temporary, seasonal, or permanent zones of SANBI was, as explained in Section 1.5, considered too vague for the purposes of discriminating between vegetation habitat units for phytoassessment and the alternative supralittoral, littoral and aquatic zones were proposed as presented in Section 1.5.2. The supralittoral and littoral zones proved to contain different sets of species when examined within each of the terrestrial units of vegetation (Section 5.2.1). These two zones are considered more easily distinguishable than the numerous hydroregime divisions proposed by SANBI (2009) (as based on temporary, seasonal, permanent levels of saturation and/or inundation) and are thus considered more useful for phytoassessment habitat classification purposes.
- Whilst numerous HGM units (*sensu* SANBI 2009) were sampled in the present study no more than one was contained within any given unit of vegetation classified to be a distinct unit based on floristic, edaphic and climatic homogeneity. The objective of testing whether the discriminators used in the National Wetland Classification System (SANBI 2009) for separating different units of wetland habitat was therefore partially

frustrated as with the present data set it was not possible to test whether different HGMs hold the same species complement.

- Geology, substrate type (sand *vs* clay), acidity *vs* alkalinity and hydrological regime (zonation) are all discriminators used by SANBI (2009) that have been shown to successfully separate different units of habitat for wetland vegetation in the present study.
- Naturalness (natural *vs* artificial), salinity and vegetation cover types (herbaceous, shrubby, forested) are discriminators that were not specifically tested with the present data set that was composed of "natural" inland freshwater wetlands dominated by herbaceous vegetation.
- The impact of unnatural (anthropogenic) disturbance influences has been shown to have considerable negative impact on wetland vegetation in this study.
- The existence of a significant climatic moisture gradient between different units of wetland vegetation suggests that this too is an essential discriminator that should be included within the SANBI (2009) National Wetland Classification Scheme. Climatic parameters of temperature range and moisture availability would provide more accurate discriminators than the ecoregions (*sensu* Kleynhans *et al.* 2005) or bioregions (*sensu* Rutherford *et al.* 2006).

This research represents the first empirical test of many of these discriminators (other than hydrological zonation) in the context of the seasonal wetlands in the Fynbos biome.

6.6.2 National wetland plant databases

Reviews of the literature suggested that there is no clarity as to which plant species indicate a given habitat or environmental condition and there is a lack of base-line data that could inform our ecological understanding of biotic and abiotic interaction in wetlands. There is limited knowledge of what biota, environmental parameters, anthropogenic disturbances or stressors and ecological conditions exist in our wetlands and a concomitant lack of understanding of the resultant ecological interactions that occur. For instance in South African wetlands, there is limited existing baseline information on:

- Plant autecological information and related physiology and tolerances, phytosociology, and community successional development; and on
- Phytogeography of comparable and distinct units of vegetation.

Ecological interpretation of such baseline data that has been gathered is currently in its infancy in this country and this current state of knowledge for South African wetland

vegetation suggests the need for collation to form a national database of existing data and baseline research.

The 'important species' listed for the terrestrial vegetation types of the Fynbos Biome in the Vegetation Atlas of Mucina and Rutherford (2006) include associated wetland taxa and soil types from which the natural range of nutrient requirements for certain wetland plants could be derived. This would require an exhaustive cross referencing review of the studies used to compile the vegetation types and their associated environmental data. If sufficiently useful environmental data was collected in these studies, then the resultant data base that such a review could provide would prove invaluable in the separation of taxa into groups based on biogeography, nutrient requirement, geological substrate types, climate and hydroregime. Such data could be used in a meta-analysis to explore wetland phytosociology.

It is apparent that considerable inventory-type information on wetland plant taxa will be required in order to be able to determine ecological indicators that can be applied locally, regionally or nationally. A vast amount of information would be gained by a phytosociological study describing species distribution and association with controlling environmental and anthropogenic variables. A Water Research Commission study is currently underway to provide this baseline data and will be informative about species that can be expected to occur under reference conditions for given habitat (WRC K5/1980 Sieben 2010). The data collated for this present Masters thesis will add to the WRC baseline dataset. Until considerable wetland autecology and phytosociology is understood, as was found in the present study, phytoassessment indices may only be able to differentiate between communities from the crudest of opposites of impaired relative to un-impacted environmental conditions and only for relatively limited phytogeographical areas.

6.6.3 Census of taxa indicative of wetness

There are many terrestrial plants in South Africa, that can be classified as 'facultative wetland' or 'facultative dry' species (*sensu* Reed 1988), that establish successfully in drained wetlands and could thus indicate dehydration of or excessive water extraction from wetlands. Some of these are indigenous and there are also many non-indigenous "ruderal" (opportunistic) terrestrial taxa. Accurate lists of what species constitute the various categories of obligate-wetland to terrestrial plants, in each region of South Africa

could assist in wetland phytoassessment. The list of taxa affiliated with South African wetlands (Rene Glen pers. com., unpublished – see Appendix 6) goes part of the way in providing this information. However, within this list (Glen unpublished) the number of observations by which species were designated as obligate or facultative wetland taxa was not included and hence the accuracy of these designations is unknown. Furthermore a number of species in Glen's unpublished list that were designated as obligate wetland species (e.g.: *Isolepis cernua* (Vahl) Roemer & Schultes var. *cernua*) have been observed to occur in ephemerally wet conditions during the course of this study and have previously been recognized as being facultative wetland species in the South African context by Cook (2004). A wetland vegetation database (as proposed above in Section 6.6.2) would assist with the endeavour to accurately categorise the affinity of species for wetland and dryland conditions. Glens unpublished list certainly provides the basis of species information for which further autecological information needs to be ascertained.

6.6.4 Indicators of functional change that result from nutrient enrichment

Research in the USA suggests that nitrogen and phosphorus are the primary nutrients limiting productivity in wetlands. In ecosystems where these nutrients are limiting, increased availability of N and P generally results in functional changes in wetlands such as increased storage of these nutrients in the tissues of wetland plants and a resultant increase in net primary productivity (NPP) (e.g. Shaver & Melillo 1984 & Shaver *et al.* 1998). Changes in nutrient uptake and NPP alter rates of uptake, storage, and release of carbon (C), N, and P, thereby affecting ecosystem process such as wetland energy and nutrient cycles, accumulation of soil organic matter and organic carbon export. Such changes can compromise wetland environmental condition by altering niche/habitat characteristics that in turn affect wetland vegetation community composition and associated faunal assemblages.

'Functional indicators' of nutrient enrichment or eutrophication include leaf N and P content and metrics of NPP in the form of biomass production and stem height. Whilst stem density also reflects increased biomass it can also reflect other factors such as vigorous clonal growth (e.g. in *Cyperus textilis*) and thus is not a reliable indicator of nutrient enrichment. Leaf nutrient contents respond most rapidly to nutrient enrichment followed by the response of stem height and increased above ground biomass (US EPA 2002d). A potential avenue for further development of metrics of environmental condition

specifically relating to nutrient enrichment of wetlands would therefore be to explore the N:P ratio in leaf material, biomass production and changes in stem heights of species/genera/life-history groups that have proved elsewhere to be useful for the determination of functional change.

6.7 Concluding remarks

Further avenues of research that will facilitate development of phytoassessment metrics with broad applicability are:

- The testing of the utility of metrics developed for one locality within other localities of similar vegetation.
- The examination of the variability of the full complement of the wetland vegetation associated with a given terrestrial vegetation unit such as the Cape Flats Sand Fynbos across the whole of the Cape Flats. Such variability may suggest that floristic differences observed in the present study between wetlands sampled from different locations (Kenilworth *vs* Lotus River) would restrict the applicability of metrics developed in one location to only be of use in that location.
- The testing of the utility of the biological indicators of functional change resulting from nutrient enrichment.
- The testing of whether different HGMs from the same geographical location (i.e. representing the same climatic and edaphic parameters) hold a different complement of species.

In the context of the remaining research objectives, this study has been enlightening about the environmental variables (or drivers) that correlate with wetland vegetation distribution and the classification of distinct units of wetland phytogeography. That wetland vegetation is not broadly and azonally distributed, (*"driven by hydroregime and salt concentrations exerting an influence greater than the macroclimate"* (Mucina *et al.* 2006a)), but determined by the same factors of climate and geology that determine the distribution of the (intra)zonal units of terrestrial vegetation is a key finding of the present study that alters our theoretical understanding of wetland plant ecology in the Fynbos biome. That climate, geology and related edaphic substrates and hydrological zonation are necessary discriminators of different units of wetland vegetation is a key to understanding how best to classify distinct vegetation units for purposes of phytoassessment in the Fynbos biome. As a result of these findings it is apparent that whilst phytoassessment has considerable national potential the speed and ease of its

development within the Fynbos biome or the *Capensis* phytochorion, and/or the spatial scale over which phytoassessment metrics will be applicable will certainly be restricted by the existence of inherent natural diversity. In less floristically and environmentally diverse areas of South Africa it might be possible to develop more widely applicable metrics.

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Appendix 1: Biotic characteristics of each wetland

Appendix 1: Biotic character of each wetland assessed in the Cape coastal lowlands including National Freshwater Ecosystem Priority Area (NFEPA*) vegetation groups.

Sub-region	Wetland	Intrazonal Vegetation Unit (Mucina *et al.* 2006a)	National Zonal Vegetation Unit (Rebelo *et al.* 2006)	Bioregion (Rutherford *et al.* 2006)	NFEPA* Groups (Roux *et al.* 2006)
West Coast	Ber01	Vernal Pool	Saldanha Flats Strandveld	West Strandveld	Western Strandveld
West Coast	Ber02	Cape Estuarine Salt Marsh	Hopefield Sand Fynbos	South West Fynbos	South West Sand Fynbos
West Coast	Vel02 in (Berg River)	Cape Lowland Freshwater	Hopefield Sand Fynbos	South West Fynbos	South West Sand Fynbos
West Coast	Dar01	Cape Lowland Freshwater	Swartland Granite Renosterveld	West Coast Renosterveld	West Coast Granite Renosterveld
West Coast	Dar01B	Cape Lowland Freshwater	Swartland Granite Renosterveld	West Coast Renosterveld	West Coast Granite Renosterveld
West Coast	Dar02	Cape Lowland Freshwater	Swartland Granite Renosterveld	West Coast Renosterveld	West Coast Granite Renosterveld
West Coast	Dar03	Lowland Alluvial	Swartland Alluvium Renosterveld	West Coast Renosterveld	West Coast Alluvium Renosterveld
West Coast	Dar05	Vernal Pool	Hopefield Sand Fynbos	West Coast Fynbos	West Coast Sand Fynbos
West Coast	Dar10	Lowland Alluvial	Swartland Alluvium Renosterveld	West Coast Renosterveld	West Coast Alluvium Renosterveld
West Coast	Dar11	Vernal Pool	Atlantis Sand Fynbos / Swartland Alluvium Renosterveld	West Coast Renosterveld	West Coast Alluvium Renosterveld
West Coast	Ver01A	Cape Lowland Freshwater	Leipoldtville Sand Fynbos	North West Fynbos	North West Sand Fynbos
West Coast	Ver01B	Cape Lowland Freshwater	Lamberts Bay Strandveld	West Strandveld	Western Strandveld
West Coast	Ver02A	Cape Lowland Freshwater	Leipoldtville Sand Fynbos	North West Fynbos	North West Sand Fynbos
West Coast	Ver02B	Cape Lowland Freshwater	Leipoldtville Sand Fynbos	North West Fynbos	North West Sand Fynbos
West Coast	Ver02C	Cape Lowland Freshwater	Leipoldtville Sand Fynbos	North West Fynbos	North West Sand Fynbos
West Coast	Ver03	Cape Lowland Freshwater	Leipoldtville Sand Fynbos	North West Fynbos	North West Sand Fynbos
Cape Flats	Dri01	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Dri03	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Dri05	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Dri06	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Dri07	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Dri08	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Dri09	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Mfu01 in Driftsands	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Mfu03 in Driftsands	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld

Cape Flats	Mfu04 in Driftsands	Vernal Pool	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Mfu05 in Driftsands	Vernal Pool	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Ken01_1	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Ken01_2	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Ken04	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Ken05	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Ken06_1	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Ken06_2	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Ken06_3	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Ken10	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Ken11	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Ken20	Cape Lowlands Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Ken21	Vernal Pool	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Lot01	Cape Lowland Freshwater	Cape Flats Sand Fynbos / Dune Strandveld	South West Fynbos or West Strandveld	South West Sand Fynbos or Western Strandveld
Cape Flats	Lot02	Cape Lowland Freshwater	Cape Flats Sand Fynbos / Dune Strandveld	South West Fynbos or West Strandveld	South West Sand Fynbos or Western Strandveld
Cape Flats	Lot03	Cape Lowland Freshwater	Cape Flats Sand Fynbos / Dune Strandveld	South West Fynbos or West Strandveld	South West Sand Fynbos or Western Strandveld
Cape Flats	Lot04	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Lot05	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Lot06	Cape Lowland Freshwater	Cape Flats Sand Fynbos	South West Fynbos	South West Sand Fynbos
Cape Flats	Lot10	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Lot11	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Lot12	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Lot13	Cape Lowland Freshwater	Cape Flats Dune Strandveld	West Strandveld	Western Strandveld
Cape Flats	Lot14	Cape Lowland Freshwater	Cape Flats Sand Fynbos / Dune Strandveld	South West Fynbos or West Strandveld	South West Sand Fynbos or Western Strandveld
Overberg	Her01	Cape Lowland Freshwater	Overberg Sandstone Fynbos	South West Fynbos	South West Sandstone Fynbos
Overberg	Her02	Cape Lowland Freshwater	Overberg Sandstone Fynbos	South West Fynbos	South West Sandstone Fynbos
Overberg	Her03	Cape Lowland Freshwater	Overberg Sandstone Fynbos	South West Fynbos	South West Sandstone Fynbos
Overberg	Mel01	Vernal Pool	Elim Ferricrete Fynbos	South Coast Fynbos	South West Ferricrete Fynbos
Overberg	Rat02	Cape Inland Salt Pan	Overberg Sandstone Fynbos	South Coast Fynbos	South West Sandstone Fynbos

Overberg	Rat03	Cape Lowland Freshwater	Overberg Sandstone Fynbos	South Coast Fynbos	South West Sandstone Fynbos
Overberg	Rat04	Cape Lowland Freshwater	Agulhas Limestone Fynbos	South Coast Fynbos	South Coast Limestone Fynbos
Overberg	Uyn01	Cape Lowland Freshwater	Elim Ferricrete Fynbos	South Coast Fynbos	South West Ferricrete Fynbos
Overberg	Was01	Cape Lowland Freshwater	Central Ruens Shale Renosterveld	East Coast Renosterveld	East Coast Shale Renosterveld
Overberg	Was02	Cape Inland Salt Pan	Central Ruens Shale Renosterveld	East Coast Renosterveld	East Coast Shale Renosterveld

Appendix 2: Abiotic characteristics of each wetland

Appendix 2: Abiotic character of each wetland assessed in the Cape coastal lowlands

Sub-region	Wetland	Long	Lat	Altitude (m)	HGM unit	Wetland Size (ha)	max depth (mm)	Hydrology Group	Hydrological regime	Hydrological zones	Drainage category	Disturbance category
West Coast	Ber01	18.205389	-32.859417	2	Depression	<0.5	0-500	Winter Inundated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
West Coast	Ber02	18.270361	-32.884917	4	Floodplain	>50	1000-1500	Winter Inundated, Summer dry	Seasonal	supralittoral, littoral & aquatic	Endorheic	impaired
West Coast	Vel02 in (Berg River)	18.239111	-32.768167	23	Depression	<0.5	0-500	Winter inundated, summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
West Coast	Dar01	18.383056	-33.373	111	Depression	0.5 to 1	500-1000	Winter Inundated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
West Coast	Dar01B	18.383056	-33.373	116	Depression	0.5 to 1	500-1000	Winter Inundated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
West Coast	Dar02	18.396222	-33.358944	116	Depression	<0.5	0-500	Winter Inundated, Summer dry	Seasonal	supralittoral	Endorheic	impaired
West Coast	Dar03	18.445194	-33.287583	93	Floodplain	0.5 to 1	0-500	Winter Inundated, Summer dry	Seasonal	supralittoral	Endorheic	impaired
West Coast	Dar05	18.396861	-33.08425	51	Depression	1 to 5	0-500	Winter inundated, Summer dry	Seasonal	supralittoral	Endorheic	reference
West Coast	Dar10	18.44675	-33.285361	49	Floodplain	0.5 to 1	0-500	Winter saturated, Summer dry	Seasonal	supralittoral	Exorheic	reference
West Coast	Dar11	18.448528	-33.283111	51	Depression	<0.5	0-500	Winter saturated, Summer dry	Seasonal	supralittoral	Endorheic	impaired
West Coast	Ver01A	18.425944	-32.331417	17	Floodplain	10 to 20	500-1000	Winter inundated, summer dry	Seasonal	supralittoral & littoral	Exorheic	impaired
West Coast	Ver01B	18.358972	-32.313417	4	Floodplain	10 to 20	500-1000	Winter inundated, summer dry	Seasonal	supralittoral & littoral	Exorheic	impaired
West Coast	Ver02A	18.470611	-32.384583	4	Floodplain	10 to 20	1000-1500	Winter inundated, summer saturated	Perennial	supralittoral, littoral & aquatic	Exorheic	impaired
West Coast	Ver02B	18.474444	-32.398	4	Channelled valley bottom	10 to 20	0-500	Winter inundated, summer saturated	Perennial	supralittoral, littoral & aquatic	Exorheic	impaired
West Coast	Ver02C	18.515139	-32.443139	4	Channelled valley bottom	10 to 20	0-500	Winter inundated, summer saturated	Seasonal	littoral & aquatic	Exorheic	impaired
West Coast	Ver03	18.539361	-32.470278	7	Channelled valley bottom	5 to 10	0-500	Winter inundated, summer dry	Seasonal	supralittoral, littoral & aquatic	Exorheic	impaired
Cape Flats	Dri01	18.663389	-34.012278	30	Depression	0.5 to 1	1000-1500	Winter inundated, Summer saturated	Perennial	supralittoral, littoral & aquatic	Endorheic	impaired
Cape Flats	Dri03	18.664389	-34.012944	30	Depression	0.5 to 1	1000-1500	Winter inundated, Summer saturated	Perennial	supralittoral, littoral & aquatic	Endorheic	impaired
Cape Flats	Dri05	18.667472	-34.011889	32	Depression	0.5 to 1	0-500	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Endorheic	reference
Cape Flats	Dri06	18.660667	-33.983694	37	Depression	1 to 5	500-1000	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Endorheic	impaired

Cape Flats	Dri07	18.659333	-33.988889	33	Depression	1 to 5	0-500	Winter saturated, Summer dry	Seasonal	supralittoral	Endorheic	impaired
Cape Flats	Dri08	18.669444	-34.012667	29	Depression	0.5 to 1	0-500	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Exorheic	impaired
Cape Flats	Dri09	18.6605	-33.985556	35	Depression	0.5 to 1	0-500	Winter saturated, Summer dry	Seasonal	supralittoral	Endorheic	impaired
Cape Flats	Mfu01 in Driftsands	18.681306	-34.012417	28	Depression	<0.5	500-1000	Winter inundated, Summer saturated	Perennial	supralittoral, littoral & aquatic	Endorheic	impaired
Cape Flats	Mfu03 in Driftsands	18.678528	-34.009639	27	Depression	1 to 5	0-500	Winter inundated, Summer saturated	Perennial	supralittoral, littoral & aquatic	Endorheic	impaired
Cape Flats	Mfu04 in Driftsands	18.679139	-34.011944	27	Depression	<0.5	0-500	Winter inundated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
Cape Flats	Mfu05 in Driftsands	18.679139	-34.011944	26	Depression	<0.5	0-500	Winter inundated, Summer dry	Seasonal	supralittoral, littoral & aquatic	Endorheic	impaired
Cape Flats	Ken01_1	18.487528	-34.009583	26	Depression	<0.5	500-1000	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Endorheic	impaired
Cape Flats	Ken01_2	18.487528	-34.009583	25	Depression	<0.5	0-500	Winter inundated, Summer saturated	Seasonal	supralittoral & littoral	Endorheic	impaired
Cape Flats	Ken04	18.488056	-34.000028	25	Depression	1 to 5	0-500	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Endorheic	impaired
Cape Flats	Ken05	18.483889	-34.00075	27	Depression	0.5 to 1	0-500	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Endorheic	impaired
Cape Flats	Ken06_1	18.492056	-34.009	30	Depression	<0.5	0-500	Winter inundated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
Cape Flats	Ken06_2	18.492056	-34.009	30	Depression	<0.5	0-500	Winter inundated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
Cape Flats	Ken06_3	18.492056	-34.009	30	Depression	<0.5	0-500	Winter inundated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
Cape Flats	Ken10	18.484806	-34.005417	26	Depression	0.5 to 1	500-1000	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Endorheic	reference
Cape Flats	Ken11	18.493944	-34.001833	27	Depression	0.5 to 1	0-500	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Endorheic	reference
Cape Flats	Ken20	18.485583	-34.006444	24	Flat	1 to 5	0-500	Winter saturated, summer dry	Seasonal	supralittoral	Endorheic	impaired
Cape Flats	Ken21	18.482361	-34.001972	27	Depression	<0.5	0-500	Winter inundated, Summer dry	Seasonal	supralittoral	Endorheic	impaired
Cape Flats	Lot01	18.505167	-34.058194	8	Depression	0.5 to 1	0-500	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Endorheic	impaired
Cape Flats	Lot02	18.499639	-34.058167	7	Depression	1 to 5	500-1000	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Endorheic	reference
Cape Flats	Lot03	18.500139	-34.057667	6	Depression	0.5 to 1	0-500	Winter inundated, Summer saturated	Perennial	supralittoral & littoral	Endorheic	impaired
Cape Flats	Lot04	18.504944	-34.053889	8	Depression	0.5 to 1	0-500	Winter inundated, Summer dry	Seasonal	supralittoral, littoral & aquatic	Endorheic	impaired
Cape Flats	Lot05	18.510417	-34.048667	9	Depression	0.5 to 1	0-500	Winter inundated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired

Cape Flats	Lot06	18.535778	-34.038111	18	Depression	1 to 5	0-500	Winter inundated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
Cape Flats	Lot10	18.498111	-34.071083	8	Depression	0.5 to 1	500-1000	Winter inundated, Summer saturated	Perennial	supralittoral, littoral & aquatic	Endorheic	reference
Cape Flats	Lot11	18.49775	-34.068111	6	Depression	0.5 to 1	500-1000	Winter inundated, Summer saturated	Perennial	supralittoral, littoral & aquatic	Endorheic	reference
Cape Flats	Lot12	18.501861	-34.067667	7	Depression	0.5 to 1	0-500	Winter inundated, Summer dry	Seasonal	supralittoral	Endorheic	reference
Cape Flats	Lot13	18.501111	-34.069889	7	Depression	<0.5	0-500	Winter inundated, Summer saturated	Perennial	supralittoral, littoral & aquatic	Endorheic	reference
Cape Flats	Lot14	18.499639	-34.058167	7	Depression	<0.5	0-500	Winter inundated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
Overberg	Her01	19.272694	-34.410944	10	Hillslope seep	1 to 5	0-500	Winter saturated, Summer saturated	Perennial	supralittoral & littoral	Exorheic	impaired
Overberg	Her02	19.249472	-34.244361	15	Hillslope seep	1 to 5	0-500	Winter saturated, Summer saturated	Perennial	supralittoral & littoral	Exorheic	impaired
Overberg	Her03	19.250111	-34.409556	18	Hillslope seep	1 to 5	0-500	Winter saturated, Summer saturated	Perennial	supralittoral & littoral	Exorheic	impaired
Overberg	Mel01	19.754444	-34.724444	32	Depression	1 to 5	0-500	Winter inundated, Summer dry	Seasonal	supralittoral, littoral & aquatic	Endorheic	impaired
Overberg	Rat02	19.697889	-34.739806	14	Flat	>50	0-500	Winter saturated, Summer dry	Seasonal	supralittoral & littoral	Endorheic	impaired
Overberg	Rat03	19.676083	-34.74225	16	Depression	20 to 50	1000-1500	Winter inundated, Summer inundated	Perennial	supralittoral, littoral & aquatic	Endorheic	reference
Overberg	Rat04	19.657361	-34.743139	22	Depression	20 to 50	500-1000	Winter inundated, Summer saturated	Perennial	supralittoral, littoral & aquatic	Endorheic	reference
Overberg	Uyn01	19.757528	-34.695972	63	Depression	<0.5	1000-1500	Winter inundated, Summer saturated	Perennial	supralittoral, littoral & aquatic	Endorheic	reference
Overberg	Was01	19.83475	-34.66425	19	Depression	>50	1000-1500	Winter inundated, Summer saturated	Perennial	supralittoral, littoral & aquatic	Exorheic	reference
Overberg	Was02	19.832972	-34.654764	18	Floodplain	>50	0-500	Winter saturated, Summer dry	Seasonal	supralittoral	Exhorheic	impaired

Appendix 3: Human Disturbance Score field sheet example

SCORE TABLE: for calculating Human Disturbance Score (HDS) at each wetland

1. Intensity of Activities

5 = Poor: currently active and major disturbance

4 = less intense than "poor", but current or active alteration

3 = active medium intensity disturbance

2 = low intensity alteration causing minor disturbance

1 = low intensity alteration or past alteration that is not currently affecting wetland

0 = natural landscape and/or no evidence of disturbance

2. Landuse characterization

Rate spatial extent: 0 = none, 1 = (< 25%), 2 = (25-50%), 3 = (50 - 90%), 4 = (>90%); then where impact exists: score as per above activity Intensity table.

Present Landuse / Activity		In wetland							Within 100m of wetland edge							Within 100 to 500m of wetland edge						
			Spatial Extent x Intensity = impact							Spatial Extent x Intensity = impact							Spatial Extent x Intensity = impact					
		Extent	WQ	WQ Impact	Hydrology	Hydro impact	Phys struc	Phys struc impact	Extent	WQ	WQ Impact	Hydro	Hydro impact	Phys struc	Phys struc impact	Extent	WQ	WQ Impact	Hydro	Hydro impact	Phys struc	Phys struc impact
Commercial afforestation																						
Agriculture - crops																						
Agriculture - livestock																						
Pugging - impact of livestock hooves																						
Agriculture - irrigation																						
Fish stocking																						
Irrigation release schemes		1	2	2	2	2	0	0	2	1	2	0	0	2	4	2	0	0		0		0
Annual pastures											0											
Perennial pastures											0											
Abandoned lands											0											
Rural development											0											
Suburban gardens									1	1	1	0	0	1	1	1	0	0		0		0
Recreational (sports field, golf estate etc.)		1	2	2	2	2	0	0	2	2	4	3	6	2	4	2	3	6		0		0
Informal settlement											0											
Urban development									1	2	2	0	0	1	1	3	0	0		0		0
Industrial											0											
Infilling		1	1	1	2	2	1	1	2	1	2	0	0	1	2	3	0	0		0		0
Mining / excavation											0											

Deep flooding (too deep for emergent veg.)													0																		
Shallow flooding													0																		
Stranded aquatic vegetation													0																		
Old high water marks													0																		
Stormwater outlets													0																		
Sewage disposal (note dist. of outflow fr. wetland)													0																		
WWTW outlets													0																		
Solid waste disposal (incl - dumping and litter)													0																		
Weirs													0																		
Berms													0																		
Dams													0																		
Water abstraction													0																		
Drainage channels													0							2	0		3		2	6			0		0
Roads / Railway										1		2	2		0		0			1	1		1		0	0			0		0
Culverts													0																		
Dredging													0																		
Pedestrian paths		1		0	0		0	0		2	2		1	2	2		1		1												
Dense woody alien vegetation patches																															
Dense aquatic alien vegetation patches																															
off road vehicle use													0																		
Erosion e.g. gullies / headcuts													0																		
Deposition / sediment													0																		
Other													0																		
SUM OF IMPACTS	**61**		**+**	**5**	**+**	**6**	**+**	**3**		**+**	**15**	**+**	**7**	**+**	**13**		**+**	**12**	**+**	**0**	**+**	**0**

3. Buffer Width vegetation categories

Width of dryland vegetation buffer		(0) unlimited and natural vegetation cover	(1) natural buffer on average > 50m	(2) buffer averages 25 - 50m from perimeter	(3) Transformed vegetation 25 – 50 meters	(4) Transformed vegetation 10 - 25m width	(5) Transformed vegetation < 10 meters	(6) No vegetated buffer
North side		**0**						
East side			**1**					
West side					3			
South side								6
Buffer Score = Ave. of four sides	**2.5**	0	1		3			6

Ken03:HDS=impacts+buffer:	**63.5**

Appendix 4: Vegetation sample field sheet

Vegetation sampling form: Wetland Research Program

Site Code: Wetland Name: Area:	Long: Lat: Altitude (m):	**Environmental Data** **HGM Type:** **Slope:**
Sample No(s). Author(s): Litter depth (cm)	Plot size (m x m): Total Cover % Open soil % Open water %	**Aspect:** **Landform:** Basin / Flat / Channel / Slope **Hydrological zone:** Aquatic /Littoral /Supralittoral **Current hydro'l condt'n:** inund'd/satur'd/moist/dry
Habitat description [1]**:** **Dominant vegetation structure** [2]**:** **Estimated Plot Roughness** [3]**:**		**Depth to water table (cm):** **Inundation depth (cm):**
Vegetation structure [4]**:**	Cover % / Av. height (m)	**Water velocity:** stagnant / slow / fast **Water fluctuation:** stable / low / high
herb:		**Drought stress:** none / temporary / seasonally
shrub:		**Soil depth:** <20cm / 0.2-1.5m / >1.5m
moss:		**Grazing/Mowing:** none, grazed/mowed, overgrazed
Presence of: mottles / organic streaks / peat / leached layer		**Burial:** none / sand / clay

Bedrock type: Shale**/** Granite / Limestone / Sandstone / Recent Sand / Leached sand / Silcrete / Laterite
Substrate type: Sand / Silt / Clay / Cobble / Boulder / Rock / Peat / Saltcrust
Soil texture: loam / sandy loam / silty loam / clay / silty clay loam / clay / sandy clay / silty clay / peat / other (specify)
Sedimentation: Erosion /balance /chemical /clastic /organic **Rockiness:** None / pebbly to bouldery / bedrock

Cover-abundance scales (Barkman *et al.* 1964)

"r":	1-2 specimens (ex)	2m:	> 100 ex, < 5%	3:	25 - 50%
"+":	3-10 ex	2a:	5 - 12.5 %	4:	50 - 75%
"1":	11-100 ex	2b:	12.5 - 25 %	5:	75 - 100%

Species (Coll. #)	Layer [4]	Cover	Field Notes:

Notes: Habitat / structure / roughness /layers

(1) e.g.: Hypersaline flat, channel, microdepression, aquatic
(2) Restioid, Cyperoid, Graminoid, Bryophyte, Woody, Bulbous, Fern
(3) 1=open soil, 2=sparse graminoid, 3=reedbed/cyperoid, 4=dense grassland, 5=Shrub graminoid combination, 6=palmiet scrub combination;
(4) moss layer, herb layer, shrub layer.

Appendix 5: Species list for the wetlands surveyed in the Cape coastal lowlands of the Fynbos Biome

Appendix 5: Species found within characteristic stands of vegetation surveyed with the Braun Blanquet method (Westhoff & van der Maarel 1978) in the wetlands of the Cape coastal lowlands of the Fynbos Biome, South Africa.

An asterisk * is indicative of an alien genus and or species to the Fynbos biome; species? = unidentifiable species; species*? = unidentifiable but alien species; p = perennial; a = annual; fa-p = facultative annual to perennial; O = obligate wetland; fw = facultative wetland; f = facultative wet or dry; fd = facultative dryland, d = dryland.
Growth forms were taken from Mucina & Rutherford (2006).

Family	Gen sp.	Growth Form	Origin	Broad Growth Form	life history	Wetland association
Aizoaceae	*Carpobrotus edulis* (L.) L.Bolus	succulent Herb	Indig	herbaceous	p	fw
Aizoaceae	*Disphyma dunsdonia* L.Bolus	succulent Herb	Indig	herbaceous	p	ow
Aizoaceae	*Drosanthemum parvifolium* (Haw.) Schwantes	succulent Herb	Indig	herbaceous	p	f
Aizoaceae	*Galenia* cf. *crystallina* (Eckl. & Zeyh.) Fenzl	succulent shrub	Indig	Woody	p	fd
Aizoaceae	*Lampranthus leptaleon* (Haw.) N.E.Br.	succulent Herb	Indig	herbaceous	p	f
Aizoaceae	species? Vygie FC 411	succulent Herb	Indig	herbaceous	p	f
Algae	*Algae*	Macroalga	Indig	algae	a	ow
Amaranthaceae	*Amaranthus* L. species*?	Herb	Alien	herbaceous	a	fd
Amaranthaceae	*Amaranthus deflexus** L.	Herb	Alien	herbaceous	p	f
Amaranthaceae	*Atriplex muelleri** Benth.	Herb	Alien	herbaceous	fa-p	fw
Amaranthaceae	*Atriplex semibaccata** R.Br.	Herb	Alien	herbaceous	fa-p	f
Amaranthaceae	*Atriplex* L. species*?	Herb	Alien	herbaceous	fa-p	f
Amaranthaceae	*Atriplex vestita* (Thunb.) Aellen	Herb	Indig	herbaceous	p	fw
Amaranthaceae	*Chenopodium album** L.	Herb	Alien	herbaceous	a	f
Amaranthaceae	*Neslia** *paniculata** subsp. *paniculata* (L.) Desv.	Herb	Alien	herbaceous	a	f
Amaranthaceae	*Sarcocornia* cf. *natalensis* (Bunge ex Ung.-Sternb.) A.J.Scott var. *affinis* (Moss) O'Callaghan	succulent Herb	Indig	herbaceous	p	fw+
Amaryllidaceae	*Carpolyza spiralis* (L'Hér.) Salisb.	Geophytic Herb	Indig	herbaceous	p	fw+
Anacardiaceae	*Searsia laevigata* (L.) F.A. Barkley var. *laevigata forma laevigata*	tall shrub	Indig	Woody	p	fw+
Anacardiaceae	*Searsia laevigata* (L.) F.A. Barkley var. *villosa* (L.f.) Moffett	tall shrub	Indig	Woody	p	fw+
Anacardiaceae	*Searsia lucida* L. (L.) F.A. Barkley *forma elliptica* (Sond.) Moffett	tall shrub	Indig	Woody	p	fw
Apiaceae	*Arctopis echinatus* L.	Herb	Indig	herbaceous	p	f
Apiaceae	*Berula erecta* (Huds.) Colville	Herb	Indig	herbaceous	a	fw+
Apiaceae	*Peucedanum cf. galbanum* (L.) Drude	Herb	Indig	herbaceous	p	fd
Apocynaceae	*Vinca cf. major** L.	Herb	Alien	herbaceous	p	fw
Aponogetonaceae	*Aponogeton angustifolius* Aiton	aquatic Herb	Indig	herbaceous	p	ow
Aponogetonaceae	*Aponogeton distachyos* L.f.	aquatic Herb	Indig	herbaceous	p	ow
Aponogetonaceae	*Aponogeton fugax* J.C.Manning & Goldblatt	aquatic Herb	Indig	herbaceous	p	ow
Araceae	*Lemna gibba* L.	aquatic Herb	Indig	herbaceous	a	ow
Araceae	*Zantedeschia aethiopica* (L.) Spreng.	Geophytic Herb	Indig	herbaceous	p	ow

Araliaceae	*Centella asiatica* (L.) Urb.	Herb	Indig	herbaceous	p	ow
Araliaceae	*Hydrocotyle verticillata* Thunb.	Herb	Indig	herbaceous	p	ow
Asparagaceae	*Asparagus capensis* L.	Herbaceous climber	Indig	herbaceous	p	f
Asparagaceae	*Asparagus declinatus* L.	Herbaceous climber	Indig	herbaceous	p	f
Asparagaceae	*Asparagus lignosus* Burm.f.	Herbaceous climber	Indig	herbaceous	p	fw
Asphodelaceae	*Bulbinella elata* P.L.Perry	succulent Herb	Indig	herbaceous	p	fw
Asphodelaceae	*Trachyandra filiformis* (Aiton) Oberm.	succulent Herb	Indig	herbaceous	p	fw+
Asphodelaceae	*Trachyandra revoluta* (L.) Kunth	succulent Herb	Indig	herbaceous	p	fw-
Asteraceae	*cf. Conyza* Less. species?	Herb	Alien	herbaceous	a	fw
Asteraceae	*Aster* L. species*?	Herb	Alien	herbaceous	fa-p	fw
Asteraceae	*Amellus asteriodes* (L.) Druce	Herb	Indig	herbaceous	p	fd
Asteraceae	*Arctotheca forbesiana* (DC.) Lewin	Herb	Indig	herbaceous	p	ow
Asteraceae	*Arctotis flaccida* Jacq.	Herb	Indig	herbaceous	fa-p	fw+
Asteraceae	*Artemesia afra* Jacq. ex Willd.	low shrub	Indig	Woody	p	fw
Asteraceae	*Chrysanthemoides monilifera* (L.) Norl.	tall shrub	Indig	Woody	p	fw-
Asteraceae	*Chrysocoma coma aurea* L.	low shrub	Indig	Woody	p	f
Asteraceae	*Cineraria geifolia* (L.) L.	Herb	Indig	herbaceous	p	f
Asteraceae	*Conyza canadensis** (L.) Cronquist	Herb	Alien	herbaceous	a	fw
Asteraceae	*Conyza scabrida* DC.	low shrub	Indig	Woody	p	fw+
Asteraceae	*Cotula coronopifolia* L.	succulent Herb	Indig	herbaceous	a	ow
Asteraceae	*Cotula pusilla* Thunb.	Herb	Indig	herbaceous	a	ow
Asteraceae	*Cotula turbinata* L.	Herb	Indig	herbaceous	a	fw
Asteraceae	*Cotula vulgaris* Levyns	Herb	Indig	herbaceous	a	ow
Asteraceae	*Elytropappus rhinocerotis* (L.f.) Less.	low shrub	Indig	Woody	p	fw+
Asteraceae	*Felicia tenella* (L.) Nees	Herb	Indig	herbaceous	a	fw
Asteraceae	*Gazania* cf. *pectinata* (Thunb.) Spreng.	Herb	Indig	herbaceous	a	fw
Asteraceae	*Gnaphalium pauciflorum* DC.	Herb	Indig	herbaceous	a	f
Asteraceae	*Gnaphalium* L. species	Herb	Indig	herbaceous	p	fw
Asteraceae	*Gorteria personata* L.	Herb	Indig	herbaceous	a	f
Asteraceae	*Haplocarpha* Less. species	Herb	Indig	herbaceous	p	f
Asteraceae	*Helichrysum cymosum* (L.) D.Don	Herb	Indig	herbaceous	p	fw
Asteraceae	*Helichrysum foetidum* (L.) Moench	Herb	Indig	herbaceous	fa-p	fw+
Asteraceae	*Helichrysum moeserianum* (ex *rutilanis*) Thell.	Herb	Indig	herbaceous	a	fw
Asteraceae	*Helichrysum patulum* (L.) D.Don	Herb	Indig	herbaceous	p	fw-
Asteraceae	*Helichrysum* Mill. seedling	Herb	Indig	herbaceous	p	f
Asteraceae	*Hippia fruitescens* (L.) L.	low shrub	Indig	Woody	p	fw
Asteraceae	*Hypochaeris radicata** L.	Herb	Alien	herbaceous	a	fw
Asteraceae	*Lactuca serriola** L.	Herb	Alien	herbaceous	fa-p	fw
Asteraceae	*Metalasia densa* (Lam.) Karis	low shrub	Indig	Woody	p	ow
Asteraceae	*Metalasia muricata* (L.) D.Don	low shrub	Indig	Woody	p	ow
Asteraceae	*Nidorella foetida* (L.) DC.	low shrub	Indig	Woody	p	fw+

Asteraceae	*Oncosiphon glabratum* (Thunb.) Kallersjö	Herb	Indig	herbaceous	a	ow
Asteraceae	*Osmitopsis asteriscoides* (P.J.Bergius) Less.	low shrub	Indig	Woody	p	ow
Asteraceae	*Plecostachys serpyllifolia* (P.J.Bergius) Hilliard & B.L.Burtt	Herb	Indig	herbaceous	p	fw+
Asteraceae	*Pseudognaphalium luteo-album** (L.) Hilliard & B.L.Burtt	Herb	Alien	herbaceous	a	fw
Asteraceae	*Pseudognaphallium undulatum* (L.) Hilliard & B.L.Burtt	Herb	Indig	herbaceous	a	fw
Asteraceae	*Rhynchopsidium sessiliflorum* (L.f.) DC.	Herb	Indig	herbaceous	a	fd
Asteraceae	*Senecio abruptus* Thunb.	Herb	Indig	herbaceous	fa	f
Asteraceae	*Senecio arenarius* Thunb.	Herb	Indig	herbaceous	a	fw-
Asteraceae	*Senecio burchellii* DC.	Herb	Indig	herbaceous	p	fw-
Asteraceae	*Senecio cf. inaequidens DC.*	Herb	Indig	herbaceous	p	f
Asteraceae	*Senecio halimifolius* L.	low shrub	Indig	Woody	p	fw+
Asteraceae	*Senecio littoreus* Thunb.	Herb	Indig	herbaceous	a	fw
Asteraceae	*Senecio rigidus* L.	low shrub	Indig	Woody	p	fw
Asteraceae	*Senecio rosmarinifolius* L.f.	low shrub	Indig	Woody	p	f
Asteraceae	*Sonchus* asper** (L.) Hill subsp. *asper*	Herb	Alien	herbaceous	a	f
Asteraceae	*Sonchus* oleraceus** L.	Herb	Alien	herbaceous	a	f
Asteraceae	*Sonchus** L. species	Herb	Alien	herbaceous	a	f
Asteraceae	*Stoebe capitata* P.J. Bergius	low shrub	Indig	Woody	p	f
Asteraceae	*Stoebe* cf. *fusca* (L.) Thunb.	low shrub	Indig	Woody	p	f
Asteraceae	*Stoebe plumosa* (L.) Thunb.	low shrub	Indig	Woody	p	fw-
Asteraceae	*Stoebe schultzii* Levyns	low shrub	Indig	Woody	p	fw
Asteraceae	*Ursinia anthemoides* (L.) Poir.	Herb	Indig	herbaceous	a	f
Asteraceae	*Ursinia nana* ssp *nana* DC.	Herb	Indig	herbaceous	fa-p	f
Asteraceae	*Ursinia tenuifolia* (L.) Poir.	Herb	Indig	herbaceous	fa-p	f
Asteraceae	*Vellereophyton dealbatum* (Thunb.) Hilliard & B.L.Burtt	Herb	Indig	herbaceous	fa-p	fw
Asteraceae	*Xanthium* strumarium** L.	low shrub	Alien	Woody	a	fw
Boraginaceae	*Myosotis arvensis** (L.) Hill	Herb	Alien	herbaceous	a	fw-
Brassicaceae	*Heliophyla meyeri* Sond. var. *meyeri*	Herb	Indig	herbaceous	a	fw+
Brassicaceae	*Sisymbrium capense* Thunb.	Herb	Indig	herbaceous	p	f
Bruniaceae	*Berzelia abrotanoides* (L.) Brogn.	low shrub	Indig	Woody	p	f
Bruniaceae	*Berzelia lanuginosa* (L.) Brogn.	low shrub	Indig	Woody	p	f
Bryopsida	Moss	moss	Indig	moss	p	fw
Bryopsida	Moss or liverwort	moss	Indig	moss	a	fw
Caryophyllaceae	*Cerastium capense* Sond.	Herb	Indig	herbaceous	a	fw
Caryophyllaceae	*Sagina apetala** Ard.	Herb	Alien	herbaceous	a	ow
Caryophyllaceae	*Silene cretica** L.	Herb	Alien	herbaceous	a	fw
Caryophyllaceae	*Spergularia media** (L.) C. Presl	succulent Herb	Alien	herbaceous	p	fw
Characeae	*Chara ecklonii* A. Braun ex Kützing	Macroalga	Indig	algae	a	ow
Characeae	*Tolypella* cf. *nidifica* (O. F. Müller) Leonhardi var. *glomerata (*Desvaux) R.D. Wood	Macroalga	Indig	algae	a	ow
Colchicaceae	*Onixotis stricta* (Burm.f.) Wijnands	Geophytic Herb	Indig	herbaceous	p	ow
Convallariaceae	*Eriospermum cordiforme* T.M.Salter	Geophytic Herb	Indig	graminoid	p	fw

Convolvulaceae	*Falkia repens* L.f.	Herb	Indig	herbaceous	p	fw+
Crassulaceae	*Crassula* cf. *vaillantii* (Willd.) Roth	succulent Herb	Indig	herbaceous	fa	ow
Crassulaceae	*Crassula glomerata* Berg.	succulent Herb	Indig	herbaceous	a	fw
Crassulaceae	*Crassula natans* Thunb.	succulent Herb	Indig	herbaceous	fa	ow
Crassulaceae	*Crassula pellucida subsp. pellucida L.*	succulent Herb	Indig	herbaceous	fa-p	fw
Cyperaceae	*Bolboschoenus maritimus* (L.) Palla	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Carex aethiopica* Schkuhr.	graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Carex cf. acutiformis* Ehrh.	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Carex clavata* Thunb.	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Carpha glomerata* (Thunb.) Nees	mega-graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Chrysitrix capensis* L.	graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Cladium mariscus* (L.) Pohl	mega-graminoid	Indig	graminoid	p	ow
Cyperaceae	*Cyperus congestus* Vahl	graminoid	Indig	graminoid	p	fw
Cyperaceae	*Cyperus fastigiatus* Rottb.	graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Cyperus sphaerospermus* Schrad.	graminoid	Indig	graminoid	p	fw
Cyperaceae	*Cyperus textilis* Thunb.	graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Cyperus thunbergii* Vahl	graminoid	Indig	graminoid	p	fw
Cyperaceae	*Eleocharis limosa* (Schrad.) Schult.	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Epischoenus gracilis* Levyns	graminoid	Indig	graminoid	p	fw
Cyperaceae	*Ficinia argyropa* Nees	graminoid	Indig	graminoid	p	f
Cyperaceae	*Ficinia capillifolia* (Schrad.) C.B. Clarke	graminoid	Indig	graminoid	p	fw
Cyperaceae	*Ficinia capitella* (Thunb.) Nees	graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Ficinia indica* (Lam.) Pfeiffer	graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Ficinia nodosa* (Rottb.) Goetgh., Muasya & D.A.Simpson	graminoid	Indig	graminoid	p	fw
Cyperaceae	*Ficinia pygmaea* Boeck.	graminoid	Indig	graminoid	p	fw
Cyperaceae	*Ficinia repens* (Nees) Kunth	graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Ficinia zeyheri* Boeck.	graminoid	Indig	graminoid	p	fw
Cyperaceae	*Fuirena hirsuta* (P. J. Bergius) P. L. Forbes	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Hellmuthia membranacea* (Thunb.) R.Haines & K.Lye	graminoid	Indig	graminoid	p	f
Cyperaceae	*Isolepis cernua* (Vahl) Roem. & Schult	graminoid	Indig	graminoid	a	fw+
Cyperaceae	*Isolepis diabolica* (Steud.) Schrad.	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Isolepis hystrix* (Thunb.) Nees	graminoid	Indig	graminoid	a	ow
Cyperaceae	*Isolepis inconspicua* (Levyns) J. Raynal	graminoid	Indig	graminoid	a	ow
Cyperaceae	*Isolepis levynsiana* Muasya & D.A.Simpson	graminoid	Indig	graminoid	a	ow
Cyperaceae	*Isolepis marginata* (Thunb.) A. Dietr.	graminoid	Indig	graminoid	a	ow
Cyperaceae	*Isolepis natans* (Thunb.) A. Dietr.	graminoid	Indig	graminoid	a	ow
Cyperaceae	*Isolepis prolifer* R. Br.	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Isolepis rubicunda* Kunth	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Isolepis sepulcralis* Steud.	graminoid	Indig	graminoid	a	ow
Cyperaceae	*Isolepis trachysperma* Nees	graminoid	Indig	graminoid	a	ow
Cyperaceae	*Isolepis venustula* Kunth	graminoid	Indig	graminoid	p	ow

Cyperaceae	*Neesenbeckia punctoria* (Vahl) Levyns	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Pycreus polystachyos* (Rottb.) P. Beauv.	graminoid	Indig	graminoid	p	fw
Cyperaceae	*Schoenoplectus* cf. *roylei* (Nees) Ovcz. & Czukav.	graminoid	Indig	graminoid	a	ow
Cyperaceae	*Schoenoplectus scirpoideus* (Schrad.) J. Browning	mega-graminoid	Indig	graminoid	p	ow
Cyperaceae	*Schoenus nigricans* L.	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Scirpoides thunbergii* (Schrad.) Sojak	graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Tetraria cuspidata* (Rottb.) C.B.Clarke var. cf. *cuspidata*	graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Tetraria* cf. *exilis* Levyns	graminoid	Indig	graminoid	p	fw+
Cyperaceae	*Tetraria crassa* Levyns or *paludosa* Levyns	graminoid	Indig	graminoid	p	ow
Cyperaceae	*Trianoptiles capensis* (Steud.) Harv.	graminoid	Indig	graminoid	a	fw+
Cyperaceae	*Trianoptiles solitaria* (C. B. Clarke) Levyns	graminoid	Indig	graminoid	a	fw+
Dennstaedtiaceae	*Histiopteris incisa* (Thunb.) J.Sm.	Herb	Indig	herbaceous	p	fw+
Dennstaedtiaceae	*Pteridium aquilinum* (L.) Kuhn	Herb	Indig	herbaceous	p	fw-
Droseraceae	*Drosera* cf. *trinervia* Spreng.	carnivorous Herb	Indig	herbaceous	p	ow
Droseraceae	*Drosera* cf. *cistiflora* L.	carnivorous Herb	Indig	herbaceous	p	ow
Ericaceae	*Erica imbricata* L.	low shrub	Indig	Woody	p	f
Ericaceae	*Erica laeta* Bartl.	low shrub	Indig	Woody	p	f
Ericaceae	*Erica margaritaceae* Sol.	low shrub	Indig	Woody	p	fw+
Ericaceae	*Erica muscosa* (Aiton) E.G.H.Oliv.	low shrub	Indig	Woody	p	ow
Ericaceae	*Erica perspicua* J.C.Wendl.	low shrub	Indig	Woody	p	fw
Ericaceae	*Erica* L. species?	low shrub	Indig	Woody	p	ow
Ericaceae	*Erica verticilata* P.J.Bergius.	low shrub	Indig	Woody	p	fw-
Ericaceae	*Erica villosa* Andrews	low shrub	Indig	Woody	p	fw
Euphorbiaceae	*Euphorbia helioscopa** L.	Herb	Alien	herbaceous	a	f
Euphorbiaceae	*Euphorbia terracina** L.	Herb	Alien	herbaceous	a	f
Fabaceae	*Acacia cyclops** A. Cunn. ex G.Don	Small tree	Alien	Woody	p	fd
Fabaceae	*Acacia saligna** (Labill.) H.L. Wendl.	Small tree	Alien	Woody	p	fd
Fabaceae	*Aspalathus sericea* P.J. Bergius	low shrub	Indig	Woody	p	fw
Fabaceae	*Aspalathus* L. species?	low shrub	Indig	Woody	p	f
Fabaceae	*Sutherlandia frutescens* (L.) R.Br.	low shrub	Indig	Woody	p	f
Fabaceae	*Lotus** *subbiflorus** Lag.	Herb	Alien	herbaceous	p	fw+
Fabaceae	*Medicago** *polymorpha** L.	Herb	Alien	herbaceous	a	f
Fabaceae	*Melilotus** *indicus** (L.) All.	Herb	Alien	herbaceous	a	f
Fabaceae	*Paraserianthes** *lopantha** (Willd.) Nielsen	Small tree	Alien	Woody	p	fw
Fabaceae	*Psoralea glaucophylla* (L.) C.H.Stirt.	Herb	Indig	herbaceous	p	fw
Fabaceae	*Psoralea monophylla* (L.) C.H.Stirt. or *laxa* T.M.Salter	low shrub	Indig	Woody	p	f
Fabaceae	*Psoralea pinnata* L.	tall shrub	Indig	Woody	p	fw+
Fabaceae	*Sesbania** *punicea** (Cav.) Benth.	tall shrub	Alien	Woody	p	fw+
Fabaceae	*Trifolium angustifolium** L. var. *angustifolium*	Herb	Alien	herbaceous	a	f
Fabaceae	*Vicia** *benghalensis** L.	Herb	Alien	herbaceous	fa-p	f
Frankeniaceae	*Frankenia repens* (P.J.Bergius) Fourc.	Herb	Indig	herbaceous	p	o

Fumariaceae	*Cysticapnos versicaria* (L.) Fedde	Herbaceous climber	Indig	herbaceous	a	fd
Fumariaceae	*Fumaria* muralis** Sond. ex Koch	Herbaceous climber	Alien	herbaceous	a	f
Gentianaceae	*Orphium fruitescens (L.) E.Mey.*	low shrub	Indig	Woody	p	fw
Gentianaceae	*Sebea albens* (L.f.) Roem. & Schult.	Herb	Indig	herbaceous	a	fw+
Gentianaceae	*Sebea exacoides* (L.) Schinz	Herb	Indig	herbaceous	a	fw+
Gentianaceae	*Sebea micrantha* (Cham. & Schltdl.) Schinz	Herb	Indig	herbaceous	a	fw+
Geraniaceae	*Geranium* cf. *molle** L.	Herb	Alien	herbaceous	a	f
Geraniaceae	*Geranium incanum* Burm.f.	Herb	Indig	herbaceous	p	f
Geraniaceae	*Geranium purpureum** N.E.Br.	Herb	Alien	herbaceous	a	f
Geraniaceae	*Geranium rotundifolium** L.	Herb	Alien	herbaceous	a	f
Geraniaceae	*Pelargonium cucullatum* (L.) L'Hér.	Herb	Indig	herbaceous	p	fw-
Geraniaceae	*Pelargonium grossularioides* (L.) L'Hér.	Herb	Indig	herbaceous	a	fw+
Geraniaceae	*Pelargonium longifolium* (Burm.f.) Jacq.	Geophytic Herb	Indig	herbaceous	p	f
Gleicheniaceae	*Gleichenia polypodioides* (L.) Sm.	Herb	Indig	herbaceous	p	fw+
Haemodoraceae	*Wachendorfia* cf. *paniculata* Burm.	Geophytic Herb	Indig	herbaceous	p	fw-
Haloragaceae	*Laurembergia repens* P.J.Bergius	Herb	Indig	herbaceous	p	ow
Hyacinthaceae	*Albuca fragrans* Jacq.	Geophytic Herb	Indig	herbaceous	p	fw
Hyacinthaceae	*Lachenalia* species	succulent Herb	Indig	herbaceous	p	f
Hyacinthaceae	*Ornithogalum* L. species?	Geophytic Herb	Indig	herbaceous	p	fw
Hypoxidaceae	*Spiloxene aquatica* (L.f.) Fourc.	Geophytic Herb	Indig	herbaceous	p	ow
Iridaceae	*Aristea glauca* Klatt	Geophytic Herb	Indig	herbaceous	p	f
Iridaceae	*Babiana tabulosa* (Burm. f.) Ker Gawl	Geophytic Herb	Indig	herbaceous	p	fw
Iridaceae	*Geissorhiza aspera* Goldblatt	Geophytic Herb	Indig	herbaceous	p	fw
Iridaceae	*Geissorhiza cf. brehmii* Eckl. ex Klatt	Geophytic Herb	Indig	herbaceous	p	fw
Iridaceae	*Geissorhiza* cf. *inflexa* (D.Delaroche) Ker Gawl	Geophytic Herb	Indig	herbaceous	p	fw
Iridaceae	*Geissorhiza imbricata* (D.Delaroche) Ker Gawl	Geophytic Herb	Indig	herbaceous	p	o
Iridaceae	*Geissorhiza ovata* (Burm.f.) Aschers & Graebn.	Geophytic Herb	Indig	herbaceous	p	fw
Iridaceae	Geophyte species?	Geophytic Herb	Indig	herbaceous	p	ow
Iridaceae	*Ixia dubia* Vent.	Geophytic Herb	Indig	herbaceous	p	f
Iridaceae	*Ixia flexuosa* L.	Geophytic Herb	Indig	herbaceous	p	fw
Iridaceae	*Micranthus alopecuroides* (L.) Rothm.	Geophytic Herb	Indig	graminoid	p	fw
Iridaceae	*Moraea cf. flaccida* Sweet	Geophytic Herb	Indig	herbaceous	p	ow
Iridaceae	*Moraea* Mill. species	Geophytic Herb	Indig	herbaceous	p	fw
Iridaceae	*Romulea* cf. *tabularis* Eckl. ex Bég.	Geophytic Herb	Indig	herbaceous	p	fw
Iridaceae	*Romulea* Maratti species?	Geophytic Herb	Indig	herbaceous	p	fw
Iridaceae	*Sparaxis bulbifera* (L.) Ker Gawl.	Geophytic Herb	Indig	herbaceous	p	fw+
Iridaceae	*Watsonia meriana* (l.) Mill. var. *meriana*	Geophytic Herb	Indig	herbaceous	p	fw+
Juncaceae	*Juncus bufonius* * L.	graminoid	Alien	graminoid	a	fw
Juncaceae	*Juncus capensis* Thunb.	graminoid	Indig	graminoid	p	fw
Juncaceae	*Juncus cephalotes* Thunb.	graminoid	Indig	graminoid	a	ow
Juncaceae	*Juncus effusus* L.	graminoid	Indig	graminoid	p	o

Juncaceae	*Juncus exsertus* Buchenau	graminoid	Indig	graminoid	p	ow
Juncaceae	*Juncus kraussii* Hochst	Graminoid	Indig	graminoid	p	o
Juncaceae	*Juncus lomatophyllus* Spreng.	Graminoid	Indig	graminoid	p	o
Juncaceae	*Juncus oxycarpus* E.Mey. ex Kunth	Graminoid	Indig	graminoid	p	ow
Juncaginaceae	*Triglochin bulbosa* L.	Geophytic Herb	Indig	herbaceous	p	ow
Lamiaceae	*Salvia africana-lutea* L.	low shrub	Indig	Woody	p	fd
Lauraceae	*Cassytha ciliolata* Nees	Epiphytic parasitic Herb	Indig	herbaceous	p	f
Lentibulariaceae	*Utricularia bisquamata* Schrank	carnivorous Herb	Indig	herbaceous	a	ow
Lentibulariaceae	*Utricularia gibba* L.	carnivorous Herb	Indig	herbaceous	fa-p	ow
Lobeliaceae	*Lobelia anceps* L.f.	Herb	Indig	herbaceous	p	fw+
Lobeliaceae	*Lobelia comosa* L.	Herb	Indig	herbaceous	p	fw
Lobeliaceae	*Lobelia erinus* L.	Herb	Indig	herbaceous	fa-p	fw
Lobeliaceae	*Monopsis debilis* (L.f.) C.Presl	Herb	Indig	herbaceous	a	fw+
Lobeliaceae	*Monopsis lutea* (L.) Urb.	Herb	Indig	herbaceous	p	fw+
Lythraceae	*Lythrum hyssopifolia* L.	Herb	Alien	herbaceous	a	ow
Malvaceae	*Malva parviflora** L. var. *parviflora*	Herb	Alien	herbaceous	p	f
Melianthaceae	*Melianthus major* L.	low shrub	Indig	Woody	p	fw
Menyanthaceae	*Nymphoides indica* (L.) Kuntze	aquatic Herb	Indig	herbaceous	p	ow
Moluginaceae	*Hypertelis trachysperma* Adamson	succulent Herb	Indig	herbaceous	a	ow
Myricaceae	*Morella quercifolia* (L.) Killick	low shrub	Indig	Woody	p	fw
Myrtaceae	*Eucalyptus conferruminata** D.J.Carr & S.G.M.Carr or cf. *camuladensis* Dehnh.	tall tree	Alien	Woody	p	f
Myrtaceae	*Leptospermum laevigatum** (Gaertn.) F.Muell.	tall shrub	Alien	Woody	p	fw
Myrtaceae	*Leptospermum* scoparium** (Gaertn.) F. Muell.	tall shrub	Alien	Woody	p	fw
Nymphaeaceae	*Nymphaea nouchali* Burm.f. var. Zanzibariensis (Casp.) Verdc.	aquatic Herb	Indig	herbaceous	fa-p	ow
Orchidaceae	*Corycium cf. orobanchoides* (L.f.) Sw.	Geophytic Herb	Indig	herbaceous	p	fw+
Orchidaceae	*Holothrix villosa* Lindl.	Geophytic Herb	Indig	herbaceous	p	fw
Orchidaceae	*species?*	Geophytic Herb	Indig	herbaceous	p	f
Oxalidaceae	*Oxalis cf. eckloniana* C. Presl	Geophytic Herb	Indig	herbaceous	p	fw+
Oxalidaceae	*Oxalis cf. luteola* Jacq.	Geophytic Herb	Indig	herbaceous	p	fw
Oxalidaceae	*Oxalis flava* L.	Geophytic Herb	Indig	herbaceous	p	fw
Oxalidaceae	*Oxalis natans* L.f.	aquatic Herb	Indig	herbaceous	p	ow
Oxalidaceae	*Oxalis pes-caprae* L.	Geophytic Herb	Indig	herbaceous	p	f
Oxalidaceae	*Oxalis* L. species?		Indig	herbaceous	p	
Oxalidaceae	Oxalis L. species?		Indig	herbaceous	p	
Oxalidaceae	*Oxalis versicolor* L.	Geophytic Herb	Indig	herbaceous	p	f
Pinaceae	*Pinus* radiata** D.Don	Tall tree	Alien	Woody	p	fw+
Plantaginaceae	*Plantago cf. crassifolia* Forssk.	Herb	Indig	herbaceous	p	f
Plantaginaceae	*Plantago lanceolata** L.	Herb	Alien	herbaceous	p	f
Plumbaginaceae	*Limonium equisetinum* (Boiss.) R.A.Dyer	Herb	Indig	herbaceous	p	fw
Plumbaginaceae	*Limonium kraussianum* (Buchinger ex Boiss.) Kuntze	Herb	Indig	herbaceous	p	fw
Poaceae	*Agrostis cf. bergiana* Trin.	graminoid	Indig	graminoid	fa-p	fw+

Poaceae	*Aira cupaniana** Guss.	graminoid	Alien	graminoid	a	f
Poaceae	*Avena fatua** L.	graminoid	Alien	graminoid	a	f
Poaceae	*Brachypodium flexum* Nees	graminoid	Indig	graminoid	p	fd
Poaceae	*Briza maxima** L.	graminoid	Alien	graminoid	a	f
Poaceae	*Briza minor** L.	graminoid	Alien	graminoid	a	f
Poaceae	*Bromus diandrus** Roth	graminoid	Alien	graminoid	a	fw
Poaceae	*Bromus hordeaceus** L.	graminoid	Alien	graminoid	a	fw
Poaceae	*Bromus pectinatus** Thunb.	graminoid	Alien	graminoid	a	fw
Poaceae	*Cortaderia* selloana** (Schult.) Aschers. & Graebn.	mega-graminoid	Alien	graminoid	p	fw
Poaceae	*Cynodon dactylon* (L.) Pers.	graminoid	Indig	graminoid	p	f
Poaceae	*Digitaria debilis** (Desf.) Willd.	graminoid	Alien	graminoid	a	fw
Poaceae	*Diplachne fusca* (L.) P.Beauv. ex Roem. & Schult.	graminoid	Indig	graminoid	p	ow
Poaceae	*Ehrharta calycina* J.E.Sm.	graminoid	Indig	graminoid	fa-p	f
Poaceae	*Ehrharta longiflora* J.E.Sm.	graminoid	Indig	graminoid	a	f
Poaceae	*Ehrharta rupestris* ssp. *dodii* Nees ex Trin.	graminoid	Indig	graminoid	p	fw
Poaceae	*Ehrharta villosa* Schult.f.	graminoid	Indig	graminoid	p	f
Poaceae	*Eragrostis curvula* (Schrad.) Nees	graminoid	Indig	graminoid	p	f
Poaceae	*Eragrostis plana* Nees	graminoid	Indig	graminoid	p	fw
Poaceae	*Eragrostis sabulosa* (Steud.) Schweick.	graminoid	Indig	graminoid	p	f
Poaceae	*Hainardia* cylindrica** (Willd.) Greuter	graminoid	Alien	graminoid	a	fw+
Poaceae	*Helictotrichon longum* (Stapf) Schweick.	graminoid	Indig	graminoid	p	f
Poaceae	*Imperatra cylindrica* (L.) Raeuschel	graminoid	Indig	graminoid	p	ow
Poaceae	*Lagurus* ovatus** L.	graminoid	Alien	graminoid	a	fw
Poaceae	*Lolium* multiflorum** Lam.	graminoid	Alien	graminoid	p	f
Poaceae	*Lolium* perenne** L.	graminoid	Alien	graminoid	p	f
Poaceae	*Merxmuellera cincta* (Nees) Conert	mega-graminoid	Indig	graminoid	p	ow
Poaceae	*Paspalum* distichum** L.	graminoid	Alien	graminoid	p	ow
Poaceae	*Paspalum* vaginatum** Sw.	graminoid	Alien	graminoid	p	ow
Poaceae	*Pennisetum clandestinum** Chiov.	graminoid	Alien	graminoid	p	fw
Poaceae	*Pennisetum macrourum* Trin.	mega-graminoid	Indig	graminoid	p	ow
Poaceae	*Pentaschistis cf. airoides (*Nees) Stapf	graminoid	Indig	graminoid	a	f
Poaceae	*Pentaschistis pallida* (Thunb.) H.P. Linder	graminoid	Indig	graminoid	p	f
Poaceae	*Pentaschistis tortuosa* (Trin.) Stapf	graminoid	Indig	graminoid	p	fw
Poaceae	*Phalaris aquatica** L.	graminoid	Alien	graminoid	p	ow
Poaceae	*Phragmites australis* (Cav.) Trin. ex Steud.	mega-graminoid	Indig	graminoid	p	ow
Poaceae	*Poa annua** L.	graminoid	Alien	graminoid	fa-p	f
Poaceae	*Polypogon monspeliensis** (L.) Desf.	graminoid	Alien	graminoid	a	fw+
Poaceae	*Polypogon strictus* Nees	graminoid	Indig	graminoid	a	ow
Poaceae	*Puccinellia* cf. *fasciculata** (Torr.) Bickn.	graminoid	Alien	graminoid	p	ow
Poaceae	*Sporobolus africanus* (Poir) Robyns & Tournay	graminoid	Indig	graminoid	p	f
Poaceae	*Sporobolus virginicus** (L.) Kunth	graminoid	Indig	graminoid	p	fw+

Poaceae	*Stenotaphrum secundatum (Walter) Kuntze*	graminoid	Indig	graminoid	p	fw+
Poaceae	*Tribolium* cf. *echinatum* (Thunb.) Renvoize	graminoid	Indig	graminoid	a	f
Poaceae	*Tribolium hispidum* (Thunb.) Renvoize	graminoid	Indig	graminoid	p	f
Poaceae	*Tribolium uniolae* (L.f.) Renvoize	graminoid	Indig	graminoid	p	f
Poaceae	*Vulpia*cf. myuros** (L.) C.C.Gmel.	graminoid	Alien	graminoid	a	fw
Polygalaceae	*Muraltia* DC. species?	low shrub	Indig	Woody	p	fw
Polygonaceae	*Persicaria attenuata* (R.Br.) Soják subsp. *africana* K.L.Wilson	Herb	Indig	herbaceous	fa-p	ow
Polygonaceae	*Persicaria decipiens* (R.Br.) K.L.Wilson	Herb	Indig	herbaceous	a	ow
Polygonaceae	*Rumex acetosella** subsp. *angiocarpus* L.	Herb	Alien	herbaceous	p	fw
Polygonaceae	*Rumex crispus** L.	Herb	Alien	herbaceous	p	fw
Polygonaceae	*Rumex lanceolatus* Thunb.	Herb	Indig	herbaceous	p	fw+
Portulacaceae	*Portulaca* oleracea** L.	succulent Herb	Alien	herbaceous	a	f
Potamogetonaceae	*Potamogeton pectinatus* L.	aquatic Herb	Indig	herbaceous	a	ow
Potamogetonaceae	*Potamogeton pusillus* L.	aquatic Herb	Indig	herbaceous	a	ow
Primulaceae	*Anagallis arvensis** L.	Herb	Alien	herbaceous	a	fw
Primulaceae	*Samolus valerandi** L.	Herb	Alien	herbaceous	fa-p	fw+
Prioniaceae	*Prionium serratum* (L.f.) Drège ex E.May.	mega-graminoid	Indig	graminoid	p	ow
Proteaceae	*Leucodendron linifolium* (Jacq.) R.Br.	tall shrub	Indig	Woody	p	ow
Resedaceae	*Reseda lutea** L. subsp. *lutea* var. *nutans* Boiss	Herb	Alien	herbaceous	fa-p	f
Restionaceae	*Calopsis viminea* (Rottb.) H.P.Linder	graminoid	Indig	graminoid	p	fw-
Restionaceae	*Elegia asperiflora* (Nees) Kunth	graminoid	Indig	graminoid	p	fw+
Restionaceae	*Elegia eqiusitacea* (Mast.) Mast.	graminoid	Indig	graminoid	p	fw+
Restionaceae	*Elegia filacea* Mast.	graminoid	Indig	graminoid	p	fw+
Restionaceae	*Elegia microcarpa* (Kunth) Moline & H.P.Linder	graminoid	Indig	graminoid	p	ow
Restionaceae	*Elegia nuda* (Rottb) Kunth	graminoid	Indig	graminoid	p	fw+
Restionaceae	*Elegia tectorum* (L.f.) Moline & H.P.Linder	graminoid	Indig	graminoid	p	fw+
Restionaceae	*Elegia verreauxii* Mast.	graminoid	Indig	graminoid	p	fw+
Restionaceae	*Ischyrolepis caespitosa* Esterh.	graminoid	Indig	graminoid	p	fw
Restionaceae	*Ischyrolepis cincinnata* (Mast.) H.P. Linder	graminoid	Indig	graminoid	p	f
Restionaceae	*Ischyrolepis paludosa* (Pillans) H.P.Linder	graminoid	Indig	graminoid	p	ow
Restionaceae	*Platycaulos compressus* (Rottb.) Linder	graminoid	Indig	graminoid	p	ow
Restionaceae	*Restio burchellii* Pillans	graminoid	Indig	graminoid	p	fw
Restionaceae	*Restio filiformis* Poir.	graminoid	Indig	graminoid	p	f
Restionaceae	*Restio quinquefarius* Nees	graminoid	Indig	graminoid	p	fw+
Restionaceae	*Restio tetragonus* Thunb.	graminoid	Indig	graminoid	p	fw+
Restionaceae	*Thamnochortus fruticosus* Berg.	graminoid	Indig	graminoid	p	fw-
Rosaceae	*Cliffortia ericifolia* L.f.	low shrub	Indig	Woody	p	fw-
Rosaceae	*Cliffortia ferruginea* L.f.	Herb	Indig	herbaceous	p	fw
Rosaceae	*Cliffortia strobilifera* Murray	tall shrub	Indig	Woody	p	fw
Rubiaceae	*Anthospermum bergianum* Cruse	low shrub	Indig	Woody	p	f
Rubiaceae	*Carpacoce spermacocea* (Rchb.f.) Sond.	Herb	Indig	herbaceous	p	fw

Rubiaceae	*Galium spurium* L.	Herb	Indig	herbaceous	a	fw
Santalaceae	*Thesium rariflorum* Sond.	low shrub	Indig	Woody	p	f
Scrophulariaceae	*Dischisma ciliatum* (P.J.Bergius) Choisy	low shrub	Indig	Woody	p	f
Scrophulariaceae	*Limosella grandiflora* Benth.	Herb	Indig	herbaceous	fa-p	ow
Scrophulariaceae	*Microdon polygaloides* (L.) Druce	low shrub	Indig	Woody	p	f
Scrophulariaceae	*Veronica anagallis-aquatica* L.	Herb	Indig	herbaceous	fa-p	fw+
Solanaceae	*Lycium ferocissimum* Miers	low shrub	Indig	Woody	p	fd+
Solanaceae	*Solanum* cf. *lycopersicum** L.	Herb	Alien	herbaceous	a	f
Solanaceae	*Solanum retroflexum* Dunal	Herb	Indig	herbaceous	a	f
Thelypteridaceae	*Thelypteris confluens* (Thunb.) C.V.Morton	Herb	Indig	herbaceous	p	ow
Thymeleaceae	*Lachnaea densiflora* Meisn.	low shrub	Indig	Woody	p	fw?
Thymeleaceae	*Lachnaea uniflora* (L.) Beyers	low shrub	Indig	Woody	p	fw
Thymeleaceae	*Passerina corymbosa Eckl. ex C.H. Wrigh t*	low shrub	Indig	Woody	p	ow
Thymeleaceae	*Passerina paludosa* Thoday	low shrub	Indig	Woody	p	fw+
Typhaceae	*Typha capensis* (Rohrb.) N.E.Br.	mega-graminoid	Indig	graminoid	p	ow
Vallerianaceae	*Valeriana capensis* Thunb.	Herb	Indig	herbaceous	p	fw+
Zannichelliaceae	*Zanichellia palustris* L.	aquatic Herb	Indig	herbaceous	fa-p	ow

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Appendix 6: Attached data CD

- Relevé data;
- Human Disturbance Score data and field sheets;
- Environmental data
- Rene Glen's list of Wetland taxa with additions for the Fynbos Biome from Goldblatt & Manning (2000).